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Aquatic Ecosystems, Water Quality, and Global Change: Challenges of Conducting Multi-stressor Global Change Vulnerability Assessments

National Center for Environmental Assessment
Office of Research and Development

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Global Change Research Program
National Center for Environmental Assessment
Office of Research and Development
U.S. Environmental Protection Agency
Washington, DC 20460

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PREFACE

This report investigates the issues and challenges associated with identifying, calculating, and mapping indicators of the relative vulnerability of water quality and aquatic ecosystems, across the United States, to the potential adverse impacts of external forces such as long-term climate and land-use change. We do not attempt a direct evaluation of the potential impacts of these global changes on ecosystems and watersheds. Rather, we begin with the assumption that a systematic evaluation of the impacts of existing stressors will be a key input to any comprehensive global change vulnerability assessment, as the impacts of global change will be expressed via often complex interaction with such stressors: through their potential to reduce overall resilience, or increase overall sensitivity, to global change. This is a well established assumption, but to date there has been relatively little exploration of the practical challenges associated with comprehensively assessing how the resilience of ecosystems and human systems in the face of global change may vary as a function of existing stresses and maladaptations. The work described in this report is a preliminary attempt to begin such an exploration.

To do so we gathered, from the literature, a set of more than 600 indicators of water quality and aquatic ecosystem condition and changes in condition, along with numerous datasets from EPA, other federal agencies, and NGOs, and we have used all of this as a testbed for identifying best practices and challenges for calculating and mapping vulnerability nationally. We investigated gaps in ideas, methods, data, and tools as well. Specifically, we explored:

- Challenges associated with identifying those indicators that speak specifically to vulnerability, as opposed to those reflecting simply a state or condition;
- Challenges associated with calculating and estimating the values of these vulnerability indicators, including establishing important indicator thresholds that reflect abrupt or large changes in the vulnerability of water quality or aquatic ecosystems;
- Challenges associated with mapping these vulnerability indicators nationally, including data availability and spatial aggregation of the data; and
- Challenges associated with combining and compositing indicators and developing multi-indicator indices of vulnerability.

We hope that this report will be a useful building block for future work on multi-stressor global change vulnerability assessments. Ultimately, we believe the work described here can contribute to bridging disconnects between the decision support needs of the water quality and aquatic ecosystem management communities and the priorities and capabilities of the global change science data and modeling communities. In addition, we hope it will help to synthesize lessons learned from more detailed, place-based, system-based, or issue-based case studies. Such studies include those conducted on individual watersheds, on wetlands, and on urban ecosystems. This synthesis will be used to obtain national-scale insights about impacts and adaptation; and to prioritize future work in developing adaptation strategies for global change impacts.

We would like to acknowledge the excellent work of the Cadmus Group, Inc. in their collaboration with NCEA to develop this draft report. In addition, a team of external expert advisors provided critical insights that have informed all of our work in the project to date: Drs. David Allan, Kathleen Miller, John Day, David Gochis, David Yates, and Thomas Meixner. Many thanks as well to Mike Slimak, whose substantial contributions greatly improved this report, as well as to our external and EPA reviewers. Finally, we would like to thank all the NCEA Global Change Research Program staff for their numerous and significant inputs to this project.

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1. INTRODUCTION

The U.S. Environmental Protection Agency (EPA) Global Change Research Program (GCRP), located within the Office of Research and Development (ORD), is a national-scale program that supports decision-making about adapting to potential climate change and other global change impacts on air and water quality, aquatic ecosystems, and human health. GCRP collaborates with EPA Program and Regional offices, and state, local, municipal, and tribal natural resource managers, to provide scientific support for these efforts. There is a large body of literature suggesting that improvements to measuring, modeling, and understanding climate changes relevant to the hydrologic cycle, water quality, and aquatic ecosystems are needed (e.g., Barsugli et al., 2009; Bates et al., 2008; Lettenmaier et al., 2008; Kundzewicz et al., 2007; Miller and Yates, 2005; Poff et al., 2002). The management strategies of the past will not necessarily be adequate given increased awareness of stressors such as climate change and land-use change. As emphasized by a number of recent publications, top-down, prediction-based assessments of the interactions between climate change and hydrologic systems, ecosystems, and human communities will likely be of limited usefulness for local decision-making. This is due to current and foreseeable limits on reducing climate uncertainties, and because these kinds of assessments are not necessarily compatible with conclusions from the social sciences about how information is used in decision-making (e.g., see Dessai et al., 2009; Johnson and Weaver, 2009; NRC, 2009; Moser and Luers, 2008; Sarewitz et al., 2000; Fischhoff, 1994).

Effective decision support will instead start with a commitment to understand the systems we manage or aim to protect and a willingness to use what we know now for decision-making, while working to learn more. In general, comparing relative vulnerabilities fits in well with this framework, because direct evaluation of the absolute effects of climate change on water quality and aquatic ecosystems is out of reach given the state of the science for many of our vulnerability indicators. Yet policy decisions must continue to be made in the absence of perfect information. Understanding the current condition of and threats posed to our environment now can be the lens through which we view the potential risks posed by global change. This can be achieved through systematic, quantitative planning frameworks that help us to understand and evaluate various management strategies across a wide range of plausible futures. The result of such planning should be the selection of management strategies that alleviate, or at least do not exacerbate, existing and anticipated vulnerabilities of water quality and aquatic ecosystems. In other words, we should seek strategies that are robust with respect to the inherent uncertainties of the problem (e.g., Brown et al., 2010; Lempert et al., 2004).

Informed by this philosophy, GCRP has developed and is implementing a multi-year research effort designed to improve national-scale understanding of the multiple complex interactions between global change and the nation's waters. Part of this work is a major effort devoted to the development of scenarios of future climate, land-use, and hydrologic change. For example, GCRP is conducting hydrologic modeling in 20 large, U.S. watersheds in an attempt to provide broad, national-scale scenarios of streamflow and nutrient/sediment loading across a wide range of potential climate and land-use changes, to improve our understanding of the plausible range of hydrologic sensitivity to global change. Such scenarios can be used, in principle, to investigate the potential negative water quality and aquatic ecosystem impacts that we must prepare to remedy, nationally, given existing and likely future vulnerabilities of our aquatic ecosystems.

But what are these existing vulnerabilities? The idea for this report began with a seemingly simple question: *How easy would it be to assess, and map, the relative vulnerability of watersheds, across a number of dimensions, for the whole United States in a meaningful, self-consistent way?* In this report, we summarize the lessons learned to date in our attempts to answer this question.

There are two main outcomes that we report on here. First, we have collected, evaluated the quality of, processed, and aggregated a large quantity of data on water quality and aquatic ecosystem indicators across the nation. Second, we have attempted to identify best practices, challenges, and gaps in ideas, methods, data, and tools for calculating and mapping vulnerability nationally. In both contexts, we hope that this report will be a useful building block for future work on multi-stressor global change vulnerability assessments.

To measure relative vulnerability, we identified indicators that reflect the three components of vulnerability as identified by the IPCC (2007a): sensitivity, exposure, and adaptive capacity. Sensitivity is the extent to which a system responds either positively or negatively to external stimuli; exposure is the degree to which a system is exposed to stressors (and in some cases, specifically climatic variations); and adaptive capacity is the ability of a system to cope with stress. Most vulnerability indicators identified in this report measure the exposure or sensitivity of water quality and aquatic ecosystems to stressors. An understanding of exposure and sensitivity may facilitate the development of adaptive capacity within a system.

It is important to clarify here that this report does not evaluate impacts of climate change on ecosystems and watersheds. Instead, it deals only with the question of how to estimate the relative effects of other, existing stressors and their potential to reduce overall resilience, or increase overall sensitivity, to climate change. It examines this question by looking at indicators of vulnerability to such stressors. We argue that a systematic evaluation of the impacts of existing stressors is a key input to any comprehensive climate change vulnerability assessment, as the impacts of climate change will be expressed via interaction with such stressors.

While the idea that existing stressors reduce resilience and increase vulnerability to climate change remains an assumption for many systems, it is an established one, deeply embedded in recent large climate change assessment efforts. For example, the IPCC 4th Assessment Working Group II report states that: "Vulnerability of ecosystems and species is partly a function of the expected rapid rate of climate change relative to the resilience of many such systems. However, multiple stressors are significant in this system, as vulnerability is also a function of human development, which has already substantially reduced the resilience of ecosystems and makes many ecosystems and species more vulnerable to climate change through blocked migration routes, fragmented habitats, reduced populations, introduction of alien species and stresses related to pollution" (IPCC, 2007a). It then goes on to provide examples from terrestrial, marine, and coastal ecosystems.

Reducing the impact of current stressors is also frequently considered to be a "no regrets" adaptation strategy for enhancing ecosystem resilience to climate change. The U.S. Climate Change Science Program (USCCSP, 2008) reviewed adaptation options for six federally managed programs in the United States: national forests, national parks, national wildlife refuges, national estuaries, marine protected areas, and wild and scenic rivers. Adaptation options were studied by reviewing available literature, data, and models, as well as by assessing the consensus within the scientific community. Decreasing current anthropogenic stresses was the adaptation approach deemed most likely to lead to good outcomes in the face of climate change uncertainties. Numerous studies confirmed that this approach was likely to be the most successful of those considered.

The idea that existing stressors reduce resilience and increase vulnerability to climate change informs both the definition of "vulnerability" that we use, and the selection of individual indicators we examine. It is key to providing the link between what these indicators measure and an understanding of the ecological and watershed impacts of climate change, and we expand upon this idea at other points in this report.

Returning to our framing question, "*How easy would it be to assess, and map, the relative vulnerability of watersheds, across a number of dimensions, for the whole United States in a meaningful, self-consistent way?*", our strategy for addressing it was as follows:

We conducted a literature search and compiled a comprehensive list of broadly defined indicators of the vulnerability of water quality or aquatic ecosystems, including those relating to ambient surface and groundwater quality, drinking water quality, ecosystem structure and function, individual species, and the provision of ecosystem services. This then formed the set of indicators for exploring a number of subsequent challenges. These challenges fall into four broad categories:

1. Challenges associated with identifying those indicators that speak specifically to vulnerability as opposed to those reflecting simply a state or condition.;
2. Challenges associated with calculating and estimating the values of these vulnerability indicators, including establishing important indicator thresholds that reflect abrupt or large changes in the vulnerability of water quality or aquatic ecosystems;
3. Challenges associated with mapping these vulnerability indicators nationally, including data availability and spatial aggregation of the data; and
4. Challenges associated with combining and compositing indicators and developing multi-indicator indices of vulnerability.

For this work, we relied on published research and on studies by EPA, other federal agencies, and well-respected institutions like the Heinz Center and the Pew Center, both for indicator definitions and for the data to support the mapping of indicators. While each study reviewed had a slightly different objective, much of the information was relevant to the goals of this project. The intent was to examine what could be accomplished with existing indicators and data sets, and for the most part we did not attempt at this point to conceive of new indicators or collect new data. As part of this work we developed a number of example maps, and we use some of these maps in this report for illustrative purposes. We recognize that approaches other than the one we took are possible, but the lessons we learned while developing strategies for compiling and mapping national-level indicator data sets under this project would likely be useful for an array of alternative approaches. This project was a starting point and its findings have broad applicability.

The next section (Section 2) briefly describes a number of EPA efforts that informed this work, and with which we could usefully integrate the ideas in this report more closely in the future. Section 3 describes the compilation and examination of the extensive set of indicators for water quality and aquatic ecosystems that was the starting point for the analyses in this report. Sections 4 through 7 then discuss the four broad categories of challenges described above. We summarize our findings and propose some recommendations in Section 8. Finally, several appendices document the following: the literature reviewed (Appendix A); the full set of more than 600 indicators initially evaluated (Appendix B); the data sources and supporting information for the 53 vulnerability indicators that were evaluated for data availability and mapping potential (Appendix C); the methodological details for how the various maps were produced (Appendix D); example maps displayed using 4-digit Hydrologic Units and their descriptions (Appendix E); example maps displayed using ecoregions and their descriptions (Appendix F); vulnerability categories for each indicator by each HUC (Appendix G); and the steps for evaluating and modifying vulnerability indicators (Appendix H).

2. SYNERGIES WITH OTHER EPA EFFORTS

There are a number of EPA efforts devoted to indicator-based assessment of environmental condition and impairment. This report draws from these efforts in a number of direct and indirect ways. In addition, greater integration of the work described here with these efforts has the potential for a number of significant benefits. Here, we briefly summarize some of these connections.

The valued role of environmental indicators in environmental resource assessment and management is evidenced in recent years by several prominent reports from both within the government sector and outside it (e.g., Heinz Center, 2008). Notably, EPA tracks roughly 83 indicators of environmental and human health for its Report on the Environment (U.S. EPA, 2008a). For example, Chapter 3 of the ROE is a report card on trends in the extent and condition of the nation's waters (U.S. EPA, 2008a). The ROE indicators are revisited roughly once every three to four months and subsequently updated online to assess changes over time. They are generally reported as national averages or representative examples, rather than as mapped distributions. The long-term goal for the ROE is to report all indicators as temporal trends. The ROE has its roots in the Environmental Monitoring and Assessment Program (U.S. EPA, 2010a), a research program within EPA's Office of Research and Development that was designed to develop the tools necessary to monitor and assess the status and trends of national ecological resources. EMAP collected field data from 1990 to 2006, and focused on developing the scientific understanding for translating environmental monitoring data from multiple spatial and temporal scales into assessments of current ecological condition and forecasts of future risks to our natural resources. We drew a number of the indicators discussed in this report, as well as general indicator definitions, from the ROE.

Monitoring of the nation's aquatic resources is now conducted by the EPA Office of Water's National Aquatic Resource Surveys (U.S. EPA, 2010b), which publishes a series of studies that report on core indicators of water condition. These studies use standardized field and lab methods that are designed to yield unbiased, statistically-representative estimates of the condition of the whole water resource, such as rivers and streams, lakes, ponds, reservoirs, and wetlands. Products of this program include the National Coastal Condition reports, the National

Wetland Condition Assessment, the Wadeable Streams Assessment, and a number of other reports. Again, as with the ROE, we drew a number of indicators from these assessments.

One of the largest and most important efforts within the agency that has relevance for indicator-based work is the Impaired Waters listing (U.S. EPA, 2010c). Section 303(d) of the Clean Water Act (CWA) requires states, territories, and authorized tribes to assess their waters and identify all water bodies (e.g., streams and rivers) that are impaired. Impaired waters are those that do not meet water quality standards because they are too polluted or otherwise degraded. Waters that do not meet state, territory, or tribal Water Quality Standards due to such impairments are placed on the CWA Section 303(d) list, scheduled for Total Maximum Daily Load (TMDL) development, and eventually restored. EPA maintains responsibility for implementing the 303(d) regulations by ensuring that impaired waters lists are developed. All impaired waters information is then provided to the public via EPA's online data system known as ATTAINS (U.S. EPA, 2010d). For this report, we considered using or developing indicators based on the 303(d) impaired waters lists from each state. Our intent was to use these lists to determine the degree to which waters are impaired for a given unit of spatial aggregation and to frame these identified impairments within a vulnerability context. This link has been previously discussed by EPA during evaluations of how water programs may need to adapt to changes in climate – e.g., EPA's National Water Program Strategy: Response to Climate Change report states that warmer air and water temperatures may lead to “increased pollutant concentrations and lower dissolved oxygen levels will result in additional waterbodies not meeting water quality standards and, therefore, being listed as impaired waters requiring a total maximum daily load (TMDL)” (U.S. EPA, 2008b, p. 9). However, we decided to forego using 303(d)-based indicators because of significant gaps in the impaired waters data, which are not comprehensive. This lack of national data is compounded by the variation in assessment programs across states. See Section 6.1.4 and Figure 6-1 for additional discussion of these issues.

EPA’s Regional Vulnerability Assessment (ReVA) program (U.S. EPA, 2009a) seeks to characterize vulnerability through investigation of ecosystem dynamics, the connectivity between ecosystems and the broader landscape, and ecosystem interactions with socioeconomic factors. The purpose of the ReVA program is to examine the probability of future problems at a regional scale, even when precise environmental conditions at a given location cannot be predicted. The ReVA program also aims to help decision-makers assess the degree and types of stress posed by human actions on a region's environmental resources. The program's methodology evaluates indicators of vulnerability, aggregates them into indices, and evaluates the likelihood of exacerbation of vulnerability as a result of future stressors. To date, the ReVA program's methodology has been applied to a comprehensive analysis of the Mid-Atlantic region (U.S. EPA, 2000a). EPA plans to conduct similar assessments in other regions.

The ReVA program is an outstanding source of vulnerability metrics and indicators. The present study complements the ReVA program by building on its extensive work on vulnerability and investigating a similar methodology for national scale investigations of vulnerability focused on climate change. Both the ReVA program and the current study present relative measures of vulnerability and identify future research opportunities that would result in measures of absolute vulnerability. Future efforts may include integration of ReVA tools and data with the indicators presented in the current report.

EPA's just-released 2010 report, *Climate Change Indicators in the United States* (U.S. EPA, 2010e), is a new effort that is intended to track and interpret a set of 24 indicators, each describing trends related to the causes and effects of climate change. It focuses primarily on the United States, but in some cases also examines global trends. EPA intends to begin using these indicators to monitor the effects and impacts of climate change in the United States, assist decision-makers on how to best use policymaking and program resources to respond to climate change, and assist EPA and its constituents in evaluating the success of their climate change efforts. We did not use these indicators in this report, but we envision integrating them with the methodologies discussed here in future efforts to assess vulnerability of water quality and aquatic ecosystems to climate change.

Finally, there is a pressing need for objective strategies to prioritize agency efforts by comparing different geographic locations in terms of their expected responses to future conditions and various management options. This can be done with regard, for example, to stream restoration (Norton et al., 2009) and to climate change adaptation (Lin and Morefield, 2011). As Norton et al. (2009) write, "Tens of thousands of 303(d)-listed waters, many with completed TMDLs, represent a restoration workload of many years. State TMDL scheduling and implementation decisions influence the choice of waters and the sequence of restoration. Strategies that compare these waters' recovery potential could optimize the gain of ecological resources by restoring promising sites earlier." Norton et al. (2009) then explore ways that states, tribes, and territories can use measurable metrics of ecological, stressor, and social context to estimate the relative recovery potential of sites, as a key input into decisions that set priorities for the selection and sequence of restoration efforts. Similarly, Lin and Morefield (2011), using the Atlantic and Gulf Coast National Estuaries as their example, propose a framework for assessing and prioritizing management recommendations that might be made in response to communities' vulnerability to climate change and their wishes to develop adaptation strategies. In our view, attention to the issues and challenges discussed in this report is likely to aid in the task of developing objective measures that can inform a broad range of prioritization decisions.

3. INDICATORS CONSIDERED FOR THIS REPORT

This section describes the approach used to compile a comprehensive list of potential indicators of water quality and aquatic ecosystem vulnerability from those identified in published sources. Figure 3-1 outlines the general methodology in the selection of indicators for this study.

3.1. LITERATURE SEARCH

We performed an extensive literature search to identify recent studies related to the monitoring and evaluation of water quality and ecosystem conditions. The types of literature reviewed included journal articles, studies, and reports. The literature ranged widely in study area, from local to international. It ranged in technical field from biological, hydrological, and chemical, to human aspects, and included both primary and secondary literature. The literature sources also varied, including individual researchers, public institutions, and non-governmental organizations. Studies reviewed spanned a decade of relevant literature from 1998 through 2008.

The literature reviewed was primarily obtained from the GCRP research team members and through internet and library database searches conducted by Cadmus. Literature identified by GCRP as relevant was considered to be “core literature” and was given high priority in the review process. Thereafter, other references were reviewed to identify additional indicators for possible inclusion. The citations within the core literature were also useful as sources of additional relevant literature.

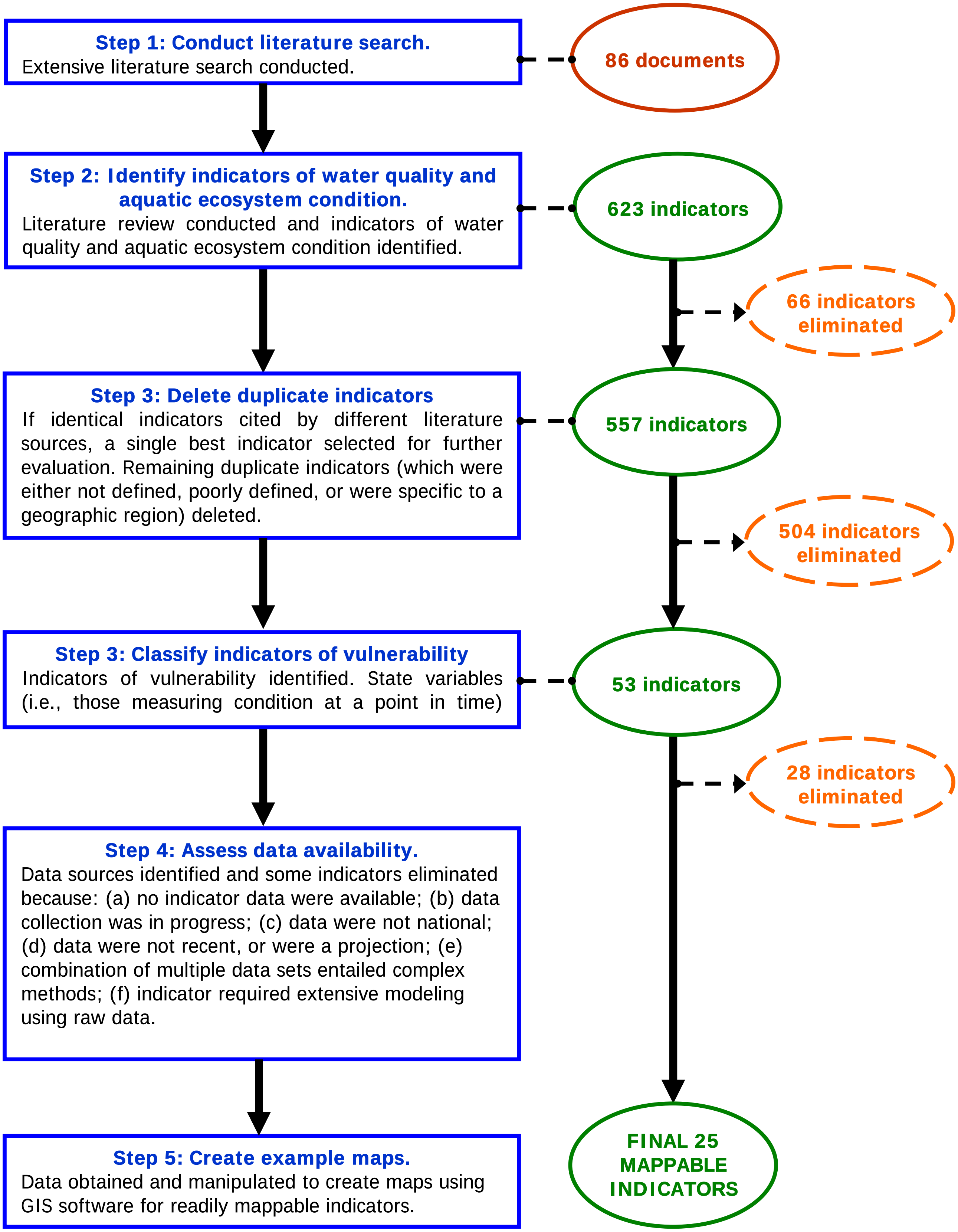
3.1.1. Core Literature

As noted above, the GCRP research team identified a short list of studies as core literature that served as a starting point for identifying vulnerability indicators. These studies are listed in Table 3-1.

Table 3-1. List of core literature

List of Core Literature (see Appendix A for full references)	
• Coastal States Organization, 2007	• Millennium Ecosystem Assessment, 2005b
• Ebi et al., 2007	• Millennium Ecosystem Assessment, 2005c
• Frumhoff et al., 2007	• National Assessment Synthesis Team, 2000a
• Gilliom et al., 2008	• National Assessment Synthesis Team, 2000b
• Gleick and Adams, 2000	• Poff et al., 2002
• Hamilton et al., 2004	• U.S. EPA, 2006
• Heinz Center, 2002	• U.S. EPA, 2008a
• Heinz Center, 2008	• U.S. EPA, 2008b
• Hurd et al., 1998	• USGAO, 2005
• Hurd et al., 1999	• United States Geologic Survey (USGS), 1999
• Lettenmaier et al., 2008	• Zogorski et al., 2006
• Millennium Ecosystem Assessment, 2005a	

Figure 3-1. Flowchart of methodology used to identify and map vulnerability indicators.



Some studies, typically those that were specifically geared towards identifying indicators of ecosystem change or documenting the results of national environmental monitoring studies, served as a source for many of the indicators in this EPA study. Some key studies in the core literature and how they were used are described below.

Hurd et al., 1998 and Hurd et al., 1999

The report, *Water Climate Change: A National Assessment of Regional Vulnerability*, prepared for EPA by Hurd et al. (1998), identified key aspects of water supply and quality that could be adversely affected by climate change, developed indicators and criteria useful for assessing the vulnerability of regional water resources to climate change, created a regional database of water-sensitive variables consistent with the vulnerability measures, and applied the criteria in a comparative national study of the vulnerability of U.S. water resources. The result of this study was a series of national-scale maps attempting to demonstrate the vulnerability of different U.S. regions to climate change for each indicator of vulnerability of water supply and quality. An abbreviated version of this study, presenting a few select indicators and outlining the general methodology used in creating national-scale maps for each indicator, was later published in the Journal of the American Water Resources Association (Hurd et al., 1999). The spatial resolution of vulnerability estimates used by Hurd et al. (1998) was a 4-digit Hydrologic Unit Code (HUC) or hydrologic subregion, of which there are 222 nationwide.

Heinz Center, 2002 and Heinz Center, 2008

The State of the Nation's Ecosystems 2008: Measuring the Land, Waters, and Living Resources of the United States prepared by the H. John Heinz Center for Science, Economics, and the Environment (hereafter referred to as the Heinz Center), was the most recent publication in an effort aimed at developing a comprehensive evaluation of the condition of the nation's ecosystems. Aspects of this effort were a model for the methodology used in the present study. We also used an older publication from the same effort (Heinz Center, 2002) to incorporate indicators that were not considered in the Heinz Center 2008 study.

The indicators in the Heinz Center reports often described the state of ecosystem attributes. Because current state was considered a component of vulnerability, the selection of these indicators typically represented the first screening step in identifying useful vulnerability indicators. The state indicators used by the Heinz Center did not explicitly describe stressors that affected those indicators, although stressors were implied for ecosystem attributes that were in a degraded state.

The Heinz Center described several indicators for which adequate data were not available. We also adopted the approach of identifying ongoing collection efforts or proposing data collection priorities for indicators of potential importance. The Heinz Center report includes terrestrial ecosystem types; the present study does not. However, the “Coasts and Oceans” and “Fresh Waters” sections of the Heinz Center report included many specific indicators that we used here.

U.S. EPA, 2006

Wadeable Streams Assessment (WSA): A Collaborative Survey of the Nation’s Streams summarizes the results of a collaborative effort led by EPA (2006) to provide a statistically defensible report on the condition of the nation’s smaller streams. Standardized methods were used to measure several physical, chemical, and biological attributes at 1,392 sites that represent the small streams in the U.S.

The database that accompanied WSA was used as a data source for mapping several of the indicators in the present study. As with some indicators from the Heinz reports, the measures reported in EPA’s WSA report (2006) reflect the current condition of the wadeable streams, rather than their specific vulnerability to future changes.

U.S. EPA, 2008a

As described in Section 2, EPA tracks roughly 83 indicators of environmental and human health, and reported on those indicators in *U.S. EPA's 2008 Report on the Environment.* The Report on the Environment (ROE) is published less frequently in hardcopy form, but continually updated online (www.epa.gov/roe). Chapter 3 of the ROE is a report card on trends in the extent and condition of the nation’s waters. The indicators in this report were generally reported as national averages or representative examples, rather than mapped distributions. Some indicators were reported as temporal trends. Indicator data were derived from multiple sources, and no new data were collected as part of this chapter. The indicators in this report are revisited roughly once every three to four months and subsequently updated online to assess changes over time. The

ROE provided several indicators for this report. Some ROE indicators of temporal trends are closely tied to the concept of vulnerability.

United State Geologic Survey (USGS), 1999

The Quality of our Nation's Waters: Nutrients and Pesticides, the first summary report from the USGS' National Water-Quality Assessment (NAWQA) program, reports on the geographic distribution, environmental drivers, and temporal trends of nutrients and pesticides in surface waters. The NAWQA data include several useful summary statistics from the broad range of physical and chemical water quality parameters measured as a part of the NAWQA program.

Under the NAWQA program, 51 sites are broken up into smaller groups that are sampled in multiple rounds (20 study units in 1991; 16 study units in 1994; and 15 study units in 1997). NAWQA is also considered the best source of information on the occurrence of pesticides in surface and groundwater. However, even with the full complement of study units (including units that were not completed at the time of the present study), the spatial coverage of NAWQA sites is relatively sparse. As with most of the literature used in the present study, NAWQA reports primarily on current condition, rather than vulnerability to future change.

3.1.2. Protocol for Collecting Additional Relevant Literature

To develop a comprehensive list of indicators cited in the published literature, an extensive and representative sample of recent studies was needed. We conducted a literature search using publicly available (e.g., Google Scholar) and non-public (e.g., ScienceDirect) search tools to identify studies with a primary or secondary focus on water quality and aquatic ecosystems. We selected studies based on their likelihood of containing water quality and aquatic ecosystem indicators.

Along with the core literature, we identified 86 studies that could be used as potential sources of indicators, including:

- 19 government reports;
- 40 peer-reviewed journal articles; and
- 27 other reports including those by non-governmental or inter-governmental organizations.

See Appendix A (List of Literature Reviewed) for a complete list of the reviewed literature.

3.2. CREATION OF A COMPREHENSIVE LIST OF INDICATORS

We reviewed the literature collected and identified indicators relevant to the present study. This section describes the guidelines we used to identify relevant indicators, and the details of the choices we made to select only certain indicators from particular studies based on these general guidelines.

We use the term, “indicator” in this report as it is commonly used in the published literature (Adger et al., 2004; Villa and McLeod, 2002; Hurd et al., 1998), to define a variable or a combination of variables that can be used to measure the change in an environmental attribute. Similar terms, such as “metric” are also widely used in the literature (Norton et al., 2009; Luers, 2005), while metric and indicator are used interchangeably in other studies (Adger, 2006; Nicholson and Jennings, 2004). For the purposes of this report, we use the terms metric and indicator interchangeably.

3.2.1. Identifying Indicators of Water Quality and Aquatic Ecosystem Condition

We reviewed all of the studies indentified in the literature search to develop a comprehensive list of indicators. Unlike a typical literature review, we reviewed these studies for indicators of water quality and aquatic ecosystem condition, rather than for their contributions to the body of knowledge on this topic. Therefore, they were reviewed for their explicit or implicit description of indicators that could potentially be used to assess the vulnerability of water quality and aquatic ecosystems to environmental change. We selected indicators following the guidelines for good indicators from EPA's Report on the Environment (ROE) as presented in Figure 3-2 (Indicator definition from EPA's 2008 Report on the Environment).

- ***Useful.*** It answers (or makes an important contribution to answering) a question in the ROE.
- ***Objective.*** It is developed and presented in an accurate, clear, complete, and unbiased manner.
- ***Data Quality.*** The underlying data are characterized by sound collection methodologies, data management systems to protect their integrity, and quality assurance procedures.
- ***Data Availability.*** Data are available to describe changes or trends, and the latest available data are timely.
- ***Representative Data.*** The data are comparable across time and space and representative of the target population. Trends depicted in this indicator accurately represent the underlying trends in the target population.
- ***Transparent and Reproducible Data.*** The specific data used and the specific assumptions, analytical methods, and statistical procedures employed are clearly stated.

Figure 3-2. Indicator definition from EPA's 2008 Report on the Environment.

This selection process resulted in a comprehensive list of 623 indicators (presented in Appendix B: Comprehensive List of Indicators). Each indicator was assigned a unique indicator identification number (Indicator ID#) – this was necessary given the large number of indicators and to avoid confusion among indicators with similar names. In subsequent sections of this report, each indicator name is associated with its parenthetical ID# (e.g., Acid Neutralizing Capacity [#1]). These identification numbers also facilitate easier referencing of each indicator in the appendices of this report.

Most water quality and aquatic ecosystem indicators found in the literature were included in the comprehensive list. However, it is important to discuss why we excluded some indicators from this list and chose not to examine them in subsequent steps of this methodology. We discuss these reasons immediately below.

3.2.2. Selection of Indicators

In the interest of thoroughness, we made broad determinations regarding whether or not each indicator, measure, or metric in a particular study could be used to characterize, evaluate, or assess water quality or aquatic ecosystems. On the rare occasions when we excluded indicators from a particular study from the comprehensive list, we documented the reasons for such exclusions – for example, indicators related to air quality were generally not considered relevant to this project, and have been well-studied elsewhere. The wide range of characteristics that describe the comprehensive list of indicators for this project can be summarized as follows:

- Indicators covered a variety of different disciplines;
- Indicators were of varying scales, from local to national;
- Indicators had varying amounts of data associated with them;
- Indicators were aggregated (made up of smaller input indicators) or disaggregated;
- Indicators were drinking water indicators or indicators related to aquatic ecosystems;
- Some were indicators related to infrastructure; and
- Indicators were potentially important to decision-makers at a variety of levels, ranging from federal, to regional and local levels.

Indicators included in the list were vetted in the literature, although to varying extents. Some studies focused solely on identifying robust water quality and ecosystem condition indicators that could be used to observe and explain changes in the natural environment. Other studies merely provided a theoretical rationale for needing the development of new indicators.

In addition to selecting specific indicators, we also reviewed the literature to obtain the following indicator-related information:

- Indicator definition, as specified in the literature, or written based on supporting text in the literature;
- Level at which it is adopted (i.e., local, state, or national);
- Whether the indicator is currently in use;
- Geographic scope (i.e., local, state, or national);
- Spatial resolution;
- Target audience (e.g., scientists, policymakers, risk analysts); and
- Rationale for the indicator's inclusion on the comprehensive list of indicators (based on information in the literature) to corroborate the indicator's relevance as an indicator of the vulnerability of waterbodies to environmental degradation.

In addition, a team of technical experts classified the potential application of each indicator to climate change as high, medium, or low. These experts, listed on page iii of this report, represent multi-disciplinary fields related to the impacts of climate change on various aspects of human life and the natural environment.

In addition to the steps described above, we took two specific actions to ensure the most comprehensive indicator list possible:

- *Creation of Indicator Categories:* Different indicators measure different aspects of potential vulnerability. By grouping like indicators, it was possible to determine which aspects of water quality and aquatic ecosystem condition were reasonably covered by the selected indicators and to identify potential coverage gaps. Therefore, to facilitate reviews of the indicator list, we established indicator categories and sub-categories, as shown in Table 3-2 (Indicator primary and secondary categories).
- *Review of Indicator List by Technical Experts:* The technical advisors reviewed a draft list of indicators and were asked to add indicators where they perceived gaps. Through this process, one indicator (Total Withdrawal Information by Source & Type of Use [#622]) was added to the comprehensive list, and a significant amount of additional detail and new information was added for the indicators already in the comprehensive list.

Table 3-2. Indicator primary and secondary categories

Ecological (161)	***Hydrological (104)***	***Chemical (96)***
• Condition of Plant Species • Distribution of Plants • Exposure to Contaminants • Habitat Condition • Non-Native Species • Species at Risk • Species Diversity • Species Populations	• Duration of Natural Events • Engineered Structures • Precipitation • Sea Level Rise • Temperature • Water Flow • Water Levels • Waves	• Carbon • Chlorophyll a • Contaminants in Sediment • Microbes • Multiple Contaminants • Nutrients • Oxygen • Pesticides • pH • Salinity • Turbidity/Clarity
Land Cover/Use (61)	***Socioeconomic (57)***	***Extreme Weather Events (16)***
• Agricultural • Coastal • Forest • Freshwater • Glaciers • Grasslands/Shrublands • Natural Cover • Urban/Suburban • Wetlands	• Housing • Policy • Recreation • Resource Use	• Drought • Fire • Flood • Storm
	Air (19)	***Soil (27)***
	• Aerosols • Ozone • Temperature	• Composition • Erosion • Sediment
	Human Populations (14)	
Other (2 [1]	• Population Size • Susceptible Populations	

[1]Note: The "Other" category has no secondary categories.

3.2.3. Exclusion of Certain Indicators and Studies

In some cases, we excluded from the comprehensive list particular indicators, groups of indicators, or all indicators from a particular study. Table 3-3 (Rationale for exclusion of certain indicators) presents the rationale for not selecting some indicators from particular studies.

3.2.4. Deletion of Duplicate Indicators

As indicators for the comprehensive list were identified from various literature sources, some redundancy was noted in some groups of indicators. When two or more indicators were identified as being very similar, one was selected to represent the group, and the others were removed from further consideration for mapping. Selected representative indicators were most often those that had a clear definition, were relevant at the national level (i.e., not limited to a small geographic region), could be quantified easily, or were obtained from this study's core literature sources. Sixty-six indicators were deleted because they were redundant with other indicators in the comprehensive list.

Table 3-3. Rationale for exclusion of certain indicators

Reasons for Exclusion of Indicators	Literature Sources (see Appendix A for full references)	
Indicators were modeled projections, specific to a non-U.S. location, or were too broadly defined.	• Arnell, 1998 • Arnell, 1999 • Barnett et al., 2005 • Bergstrom et al., 2001 • Conway and Hulme, 1996 • de Wit and Stankiewicz, 2006	• Gleick and Adams, 2000 • Kundzewicz et al., 2008 • Lettenmaier et al., 2008 • Nicholls and Hoozemans, 1996 • Palmer et al., 2008 • Roderick and Farquhar, 2002
Indicators were of human adaptive capacity or socioeconomic indicators, rather than of aquatic ecosystems or water quality.	• Adger at al., 2004 • Brooks et al., 2005 • Ebi et al., 2007 • Frumhoff et al., 2006 • Frumhoff at al., 2007 • Gleick and Adams, 2000 • Jacobs et al., 2000	• Kling et al., 2003 • Millennium Ecosystem Assessment, 2005a • Millennium Ecosystem Assessment, 2005b • Twilley et al., 2001
Indicators were identical or very similar to those in another study, or indicators were better defined in another study.	• Bradbury et al., 2002 • Bunn and Arthington, 2002 • Chesapeake Bay Program, 2008 • Dai et al., 1999 • Frumhoff et al., 2007 • Grimm et al., 1997 • Hamilton et al., 2004 • Hayslip et al., 2006	• Huntington et al., 2004 • Hurd et al., 1998 • Kling et al., 2003 • Long Island Sound Study, 2008 • Ojima et al., 1999 • U.S. EPA, 1995 • U.S. EPA, 2002 • Zogorski et al., 2006
Indicators and their associated data sources were not adequately detailed as the study was primarily a policy/funding-oriented document.	• Coastal States Organization, 2007 • Luers et al., 2006 • Murdoch et al., 1999 • National Assessment Synthesis Team, 2000b • Poff et al., 2002 • U.S. EPA, 2008c	• USGAO, 2000 • USGAO, 2002 • USGAO, 2004 • USGAO, 2005 • Vincent and Pienitz, 2006 • Yamin et al., 2005
Indicators were large aggregates of smaller indicators.	• Gleick and Adams, 2000	• U.S. EPA, 2008d

4. CHALLENGES PART I: INDICATOR CLASSIFICATION

This section describes how we evaluated the indicators introduced in the previous section to determine whether they were suitable, in principle, for assessing relative vulnerability to large-scale environmental degradation due to external stressors (of which climate change would be one example). First we considered how to define vulnerability. We then applied that definition to each of the 623 indicators that resulted from the process described in the previous section, resulting in a small subset being classified as “vulnerability” indicators.

4.1. DEFINING VULNERABILITY

There has been considerable debate in the literature on the meaning of vulnerability in the context of environmental systems and stressors (climate change in particular) and the elements of which it is composed. We summarize some of that discussion here as background.

It has been argued that the lack of a common definition has hindered interdisciplinary discourse on the topic and the development of a common framework for vulnerability assessments (Füssel, 2007; Brooks, 2003). Others have argued that the purpose of the analysis should guide the selection of the most effective definition or conceptualization (Kelly and Adger, 2000).

Some of the purposes for which climate change vulnerability assessments may be performed include: increasing the scientific understanding of climate-sensitive systems under changing climate conditions; informing the specification of targets for the mitigation of climate change; prioritizing political and research efforts to particularly vulnerable sectors and regions; and developing adaptation strategies that reduce climate-sensitive risks independent of their attribution. Each of these purposes has specific information needs and thus might require a targeted approach to provide this information.

Below is a summary of discussions about the definition of vulnerability in the literature on climate change, including:

- Determinants of vulnerability;
- Defining a vulnerable situation;

- Biophysical and socioeconomic domains; and
- Predictability and uncertainty.

4.1.1. Determinants of Vulnerability

The IPCC definition of vulnerability is: “The degree to which a system is susceptible to, or unable to cope with, adverse effects of climate change, including climate variability and extremes. Vulnerability is a function of the character, magnitude, and rate of climate variation to which a system is exposed, its sensitivity, and its adaptive capacity.” (IPCC, 2007a, p. 995) (IPCC Def. 1). Three terms are defined further in the IPCC report: sensitivity, exposure, and adaptive capacity.

The IPCC defines sensitivity as “the degree to which a system is affected, either adversely or beneficially, by climate-related stimuli.” This definition is generally supported by much of the literature on the topic, but there are two subtly different interpretations. The first considers sensitivity as the probability or likelihood of passing a critical threshold in a variable of interest (e.g., the probability of exhausting water supplies) (Fraser, 2003; Jones, 2001). The second considers sensitivity to be the degree to which outputs or attributes change in response to changes in climate inputs (Moss et al., 2001). This second interpretation incorporates an understanding that some stresses may increase gradually, instead of emphasizing the passing of one critical threshold value as the only kind of important change. In both cases, a system’s sensitivity to stress is separate from its exposure to stress.

Similarly, exposure is “The nature and degree to which a system is exposed to significant climatic variations.” A system may be currently exposed (or predicted to be exposed in the future) to significant climatic variations. Because there are multiple factors related to climate and climate change that may cause stress (e.g., temperature, precipitation, winds, changes in spatial and temporal variability and extremes, etc.), the type of exposure (“hazard” in Füssel’s [2007] terminology) should be specified. In this definition, exposure is separate from sensitivity. A system may be exposed to significant climate changes, but if it is not sensitive to those changes, it is not vulnerable. The socioeconomic literature on vulnerability tends to lump these factors together (e.g., “Social vulnerability to climate change is defined as the exposure of groups or individuals to stress as a result of the impacts of climate change” [Adger, 1999]).

Finally, adaptive capacity is "The ability of a system to adjust to climate change (including climate variability and extremes) to moderate potential damages, to take advantage of opportunities, or to cope with the consequences." In the socioeconomic literature, vulnerability is often defined primarily by adaptive capacity, particularly as it is linked to poverty (e.g., "…the vulnerability of any individual or social grouping to some particular form of natural hazard is determined primarily by their existent state, that is, by their capacity to respond to that hazard, rather than by what may or may not happen in the future." Kelly and Adger, 2000; see also Olmos, 2001; and Tompkins and Adger, 2004). This conceptualization views sensitivity to most hazards as a given, exposure to some hazard(s) as inevitable, and therefore the need for adaptation will arrive sooner or later. Other authors have argued that because adaptive capacity is not necessarily static (i.e., it can be developed), vulnerability assessments should focus on sensitivity and exposure, with the goal of identifying locations to focus the development of adaptive strategies (O'Brien et al., 2004; Kelly and Adger, 2000).

4.1.2. Defining a Vulnerable Situation

There is general agreement in the literature that the term, "vulnerability," by itself, may not be sufficiently descriptive (Moreno and Becken, 2009; Füssel, 2007; Polsky et al., 2007; Brooks, 2003). Instead, a *vulnerable situation* should be defined. This definition should include the following components (Füssel, 2007):

- **Temporal reference:** the point in time or time period of interest. Specifying a temporal reference is particularly important when the risk to a system is expected to change significantly during the time horizon of a vulnerability assessment, such as for long-term estimates of climate change.
- **Sphere:** Internal (or 'endogenous' or 'in place') vulnerability factors refer to properties of the vulnerable system or community itself, whereas external (or 'exogenous' or 'beyond place') vulnerability factors refer to something outside the vulnerable system that adds to the vulnerability of the system.
- **Knowledge domain:** socioeconomic (e.g., poverty) vs. biophysical (e.g., flow regime sustainability).
- **System:** the system of analysis, such as a coupled human–environment system, a population group, an economic sector, a geographical region, or a natural system.

- **Attribute of concern:** the valued attributes of the vulnerable system that are threatened by its exposure to a hazard. Examples of attributes of concern include human lives and health; the existence, income and cultural identity of a community; and the biodiversity, carbon sequestration potential, and timber productivity of a forest ecosystem.

- **Hazard:** a potentially damaging physical event, phenomenon, or human activity that may cause the loss of life or injury, property damage, social and economic disruption, or environmental degradation.

An example of a fully specified vulnerable situation is: 'vulnerability of the incomes of the residents of a specific watershed to drought'. In practice, only the components of the definition that are not clear from the context (or uniformly applied to multiple situations) need be defined. The advantage of a specific definition of a vulnerable situation is that it is unambiguous. The disadvantage is that it makes it difficult to conduct holistic vulnerability comparisons among locations.

4.1.3. Biophysical and Socioeconomic Domains

In the climate change literature, the term "vulnerability" has more frequently been applied to socioeconomic situations; the term "risk" has been used to describe biophysical condition situations (e.g., Jones, 2001). Biophysical vulnerability or risk is primarily related to sensitivity and exposure, while socioeconomic vulnerability is more a function of adaptive capacity. Biophysical vulnerability may encompass effects on humans, such as increase in population at risk of flooding due to sea level rise. However, it is related to human exposure to hazard rather than to the ability of people to cope with hazards once they occur (Brooks, 2003). The view of vulnerability as a state (i.e., as a variable describing the internal state of a system) has arisen from studies of the structural factors that make human societies and communities susceptible to damage from external hazards. Social vulnerability encompasses all those properties of a system independent of the hazards to which it is exposed that mediate the outcome of a hazardous event (Brooks, 2003). In theory, this idea could be applied to biophysical systems, inasmuch as previous stress has rendered the system more susceptible to any new hazard.

Most of what we define as "vulnerability indicators" in this report are biophysical indicators. They therefore primarily encompass sensitivity and exposure to environmental stresses. Adaptive capacity can be developed in locations that are sensitive and exposed to stress. In addition, while much of the literature on ecosystem vulnerability, particularly as it relates to climate change, focuses exclusively on the degradation of ecosystem components that directly serve human needs (Füssel, 2007), several of the indicators in this report focus on the direct, inherent vulnerability of the aquatic ecosystems themselves, independent of the ecosystem services provided to humans. We also examine other indicators that focus on the vulnerability of drinking water quality, and are thus more obviously and directly related to human needs.

4.1.4. Predictability and Uncertainty

The future behavior of socio-ecological systems is difficult, or perhaps impossible, to predict because the components of these systems are constantly adapting to changing conditions. As a result, a system may contain non-linearities, inter-dependencies, and feedback loops that make its overall behavior unpredictable (Moreno and Becken, 2009; Fraser et al., 2003; Holling, 2001). A vulnerability assessment itself may reduce future vulnerabilities by helping target the development of adaptive capacity in systems that are sensitive and exposed to external stressors such as climate change.

For climate change in particular, many of the adverse effects on ecosystems and human systems are expected to occur as a result of stochastic events that may or may not happen, but to which a subjective probability of occurrence could in principle be assigned. Because these probabilities are conditioned on, for example, predictions of future climate and on models of how the system will respond to climate changes (Jones et al., 2001), it may not be possible to constrain them very much given the current limitations of climate prediction, as discussed in the Introduction. This report focuses on the challenges associated with assessing vulnerability across the nation without depending on accurate environmental prediction. That is, for most of the report we evaluate the vulnerability of water quality and aquatic ecosystems in the absence of specific future scenarios of global climate, population, and land use changes. This bottom-up approach of focusing on indicators vetted in the scientific literature, available data, and current vulnerability, can be used in follow-up studies in combination with approaches focused on improving our ability to predict environmental changes.

4.2. CLASSIFYING VULNERABILITY INDICATORS

In the early phases of this project, we held a workshop[1] to develop rules of thumb for classifying the comprehensive suite of 623 indicators into two broad categories. The first category is "vulnerability indicators" that, at least in principle, could measure the degree to which the resource being considered (e.g., watershed, ecosystem, human population) is susceptible to, and unable to cope with, adverse effects of externally forced change. Such change could potentially include climate or any other global change stressor. The second category constitutes state variables or indicators of condition that merely measure the current state of a resource without relating it to vulnerability.

Informed by the literature above, the workshop participants concluded that, in practical terms, to qualify as a measure of "vulnerability," an indicator should inherently include some relative or value judgment. Examples include comparing one watershed to another, comparing the indicator to some objectively defined threshold or possible state, or reporting on the indicator's change over time. Measures of water quality or ecological condition at a point in time without reference to a baseline would not make good vulnerability indicators. Viewed from the perspective of indicator measurement, this can be achieved by such methods as computing a ratio of two quantities, at least one of which is a time rate of change or a measure of variation, or computing the portion of a distribution that lies above or below a defined threshold. Examples abound, including the ratio of the standard deviation of annual streamflow to mean annual streamflow (to measure degree of variability in the stream), the ratio of stream withdrawals of water to mean annual streamflow (to measure the portion of the flow that is being used), the ratio of mean annual baseflow to mean annual total flow (to measure the susceptibility to dry periods), and the average number of days in a year that a metric such as temperature, dissolved oxygen, or salinity in coastal wetlands exceeds a particular threshold.

Applying these rules of thumb is straightforward for some of the indicators and less so for others. Many could arguably fall into either the "vulnerability" or the "state" category. For example, when assessing vulnerability to flooding, we might examine the total number of people

[1]The workshop took place at the National Center for Environmental Assessment (NCEA), in Washington, DC, on December 18, 2008. Participants included members of the Cadmus team, members of the EPA Global Change Research Program (GCRP) staff from NCEA, and the outside expert consultants acknowledged in this report.

living within the 100- or 500-year floodplain in a given watershed; when measuring ecosystem health, we might look at the total number of species in each watershed classified as "at risk." The key for these examples is that, by embedding an implied threshold in these indicators – i.e., by choosing the particular flood frequency (e.g., 100-year or 500-year) that we consider to be damaging, or a particular classification of "at risk" – we have made a judgment about the system that goes beyond assessing its condition to assessing its susceptibility to harm. Not all vulnerability indicators incorporate implied thresholds, and those that vary over a gradual gradient are still of great value and can inform assessments of relative vulnerability, as discussed in Section 5.1.

This classification exercise winnowed the original list of 623 indicators down to 53 indicators shown in Table 4-1 (List of vulnerability indicators). Examples illustrating these classification principles include the following:

Vulnerability Indicators:

- Stream Habitat Quality (#284) – compares stream habitat conditions in a given area to those in a relatively undisturbed habitat in a similar ecosystem;
- Groundwater Depletion (#121) – compares the average groundwater withdrawals to annual average baseflow, reflecting the extent to which groundwater use rates may be exceeding recharge; or
- Wetland Species At-Risk (#326) – examines the number of threatened and endangered species inhabiting a particular wetland area.

State Variables:

- Nitrogen and Phosphorus - large rivers (#186) – measurement of nitrogen and phosphorus in all streams without a reference value; or
- Instream fish habitat (#138) – a measure of instream fish concealment features (e.g., undercut banks, boulders, large pieces of wood, brush) within a stream and along its banks, without specifying reference conditions, such as, for example, concealment features at undisturbed sites.

Table 4-1. List of vulnerability indicators

Indicator (See Appendix B for definitions)	Literature Source (See Appendix A for full citations)
Acid Neutralizing Capacity (ANC) (#1)	U.S. EPA, 2006
Altered Freshwater Ecosystems (percent miles changed) (#17)	Heinz Center, 2008
At-Risk Freshwater Plant Communities (#22)	Heinz Center, 2008
At-Risk Native Freshwater Species (#24)	Heinz Center, 2008
At-Risk native marine species (relative risk) (#27)	Heinz Center, 2008
Coastal Vulnerability Index (to sea level rise) - CVI (#51)	Day et al., 2005
Commercially important fish stocks (size) (#55)	Heinz Center, 2008
Fish and Bottom-Dwelling Animals (comparison to baseline) (#95)	Heinz Center, 2008
Flood events (frequency) (#100)	Lettenmaier et al., 2008
Freshwater Rivers and Streams with Low Index of Biological Integrity (ecosystem condition) (#116)	Heinz Center, 2008
Groundwater Depletion - Ratio of Withdrawals/ Baseflow (#121)	Hurd et al., 1998
Groundwater reliance (#125)	Hurd et al., 1998
Harmful algal blooms (occurrence) (#127)	Heinz Center, 2008
Invasive species - Coasts affected (area, ecosystem condition) (#145)	Heinz Center, 2008
Invasive species in estuaries (percent influenced) (#149)	Heinz Center, 2008
Low flow sensitivity (mean baseflow) (#159)	Hurd et al., 1998
Meteorological drought indices (#165)	Jacobs et al., 2000
Number of Dry Periods in Grassland/Shrubland Streams and Rivers (Percent of streams with dry periods over time) (#190)	Heinz Center, 2008
Ratio of Snow to Precipitation (S/P) (#218)	Lettenmaier et al., 2008
Ratio of water withdrawals to annual streamflow (level of development) (#219)	Hurd et al., 1998
Riparian Condition (Riparian Condition Index) (#231)	Heinz Center, 2008
Status of Animal Communities in Urban and Suburban Streams (Percent of urban/suburban sites with undisturbed and disturbed species) (#276)	Heinz Center, 2008
Streamflow variability (annual) (#279)	Hurd et al., 1998
Stream habitat quality (#284)	Heinz Center, 2008
Water Clarity Index (real vs. reference) (#318)	NEP, 2006
Water Quality Index (5 components) (#319)	NEP, 2006
Waterborne human disease outbreaks (events) (#322)	Heinz Center, 2008
Wetland loss (#325)	MEA, 2005

Indicator (See Appendix B for definitions)	Literature Source (See Appendix A for full citations)
Wetland and freshwater species at risk (number of species) (#326)	Hurd et al., 1998
Ratio of water use to safe yield (#328)	Schmitt et al, 2008
Erosion rate (#348)	Murdoch et al., 2000
Instream use/total streamflow (#351)	Meyer et al., 1999
Total use/total streamflow (#352)	Meyer et al., 1999
Snowmelt reliance (#361)	IPCC, 2007
Pesticide toxicity index (#364)	USGS, 2006
Population Susceptible to Flood Risk (#209)	Hurd et al., 1998
Herbicide concentrations in streams (#367)	USGS, 1999
Insecticide concentrations in streams (#369)	USGS, 1999
Organochlorines in Bed Sediment (#371)	USGS, 1999
Herbicides in Groundwater (#373)	USGS, 1999
Insecticides in Groundwater (#374)	USGS, 1999
Salinity intrusion (coastal wetlands) (#391)	Poff et al., 2002
Heat-Related Illnesses Incidence (#392)	Pew Center, 2007
Precipitation Elasticity of Streamflow (#437)	Sankarasubramanian et al., 2001
Ratio of reservoir storage to mean annual runoff (#449)	Lettenmaier et al., 2008
Runoff Variability (#453)	Lettenmaier et al., 2008
Macroinvertebrate Index of Biotic Condition (#460)	U.S. EPA, 2006
Macroinvertebrate Observed/Expected (O/E) Ratio of Taxa Loss (#461)	U.S. EPA, 2006
Coastal Benthic Communities (#462)	U.S. EPA, 2008
Threatened & Endangered Plant Species (#467)	U.S. EPA, 2008
Vegetation Indices of Biotic Integrity (IBI) (#475)	U.S. EPA, 2008
Instream Connectivity (#620)	Heinz Center, 2008
Water Availability: Net Streamflow per capita (#623)	Hurd et al., 1998

All of the indicators listed in Table 4-1 were further examined for data availability and mappability, as discussed in detail in Section 6.

4.3. HOW DO THESE INDICATORS REFLECT VULNERABILITY?

All of the 53 vulnerability indicators vary in their responses to environmental stress and in the degrees to which they reflect vulnerability of water quality and aquatic ecosystems. Here we discuss, for the subset of 25 vulnerability indicators that were mappable at the national scale, how the literature characterizes the link between each indicator and the potential vulnerability of ecosystems or human systems.

Acid Neutralizing Capacity (#1)

Definition: The Acid Neutralizing Capacity or ANC (#1) indicator is a measure of the ability of stream water to buffer acidic inputs (U.S. EPA, 2006). Streams may be naturally acidic due to the presence of dissolved organic compounds (U.S. EPA, 2006). However, acid deposition arising from anthropogenic sources may increase the acidity of the stream (U.S. EPA, 2006). Acid mine drainage, formed by water passing through mines and mine tailings, is the primary source of acid in surface water, and results in the formation of concentrated sulfuric acid. Acidity is also caused by acid rain formed by dissolution of industrial and automotive emissions, such as nitrogen oxide and sulfur dioxide, in rain water (U.S. EPA, 2006).

Measurement/Calculation: The ANC indicator is calculated as the percent of stream sites that have been deemed to be at risk, i.e., that have ANC values of 100 milliequivalents (a baseline condition) or less. The data used to map this indicator were collected every five years.

Impacts of global climate change and other stressors: Acid deposition from anthropogenic sources may lower the pH of a stream with low ANC, thereby affecting aquatic vegetation and organisms, as well as water quality, particularly in sensitive watersheds. Changes in precipitation due to global climate change may result in increased acid deposition or drainage from acid mines. Areas with a low percentage of streams with suitable buffering capacity could experience disproportionately large adverse effects resulting from increased acid exposure. In contrast, well-buffered streams with higher ANC may not be as sensitive to increased acidity from external sources.

At-Risk Freshwater Plant Communities (#22)

Definition: This indicator describes the risk of elimination faced by wetland and riparian plant communities. The condition of these communities is considered important because of the ecosystem services they provide, including habitat for a variety of species, flood storage, water quality improvements, carbon storage, and other benefits (Heinz Center, 2008; Johnson et al., 2007; NRC, 1992). Loss of community types reduces ecological diversity and may eliminate habitat for rare and endangered species. At-risk status is a vulnerability indicator for aquatic ecosystems by definition, identifying communities that may have less resistance to stressors because they are already compromised.

Measurement/Calculation: Identifying which communities are at risk and their degree of endangerment is useful for planning conservation measures (Grossman et al., 1998). The Heinz Center (2008) describes three risk categories: vulnerable (moderate risk), imperiled (high risk), and critically imperiled (very high risk). Factors that were used to assign these risk categories include range, the number of occurrences, whether steep declines have occurred, and other threats.

Impacts of global climate change and other stressors: A number of environmental changes might alter the risk status of a plant community. Changes in land use and climate-related changes may decrease the range of a given plant community. The ranges of some plants may shift with temperature changes. Drying would reduce the ranges of some plants, but increased precipitation may allow some species to expand their ranges. Sea level rise associated with global climate change or a reduction in the input of freshwater may allow drought-resistant or salt-resistant plants to move into areas once dominated by freshwater plants (Lucier et al., 2006). Many potential effects on at-risk freshwater plant communities are poorly understood, including alterations in biogeochemical cycling and the effects of increased severity of storms.

At-risk Native Freshwater Species (#24)

Definition: Similar to the previous entry, this indicator describes the risk of extinction faced by 4,100 native freshwater species, including fish, aquatic mammals, aquatic birds, reptiles and amphibians, mussels, snails; crayfishes, shrimp, and insects (Heinz Center, 2008). Plants are not included. The status of these species is important because of their value both individually (e.g., as food or for other purposes) and as part of aquatic ecosystems. The at-risk status assigned to these species again directly reflects vulnerability, identifying organisms that may have less resistance to stressors because they are already compromised and have experienced a decline; further declines for some may result in extreme rarity or even extinction.

Measurement/Calculation: The Heinz Center (2008) describes four risk categories: vulnerable, imperiled, critically imperiled, and extinct. Assignment to the “vulnerable,” “imperiled,” and “critically imperiled” categories is based on up to twelve factors, including population size, number of populations, range, steep or widespread decline, or other evidence of risk.

Impacts of global climate change and other stressors: A number of external stressors might affect risk category. For example, changes in the hydrologic cycle, whether induced by climate or land-use change, may reduce available habitat and alter the range and number of locations where species occur. Sea level rise may flood freshwater habitats. Degradation of water quality and presence of certain contaminants may affect the health and long-term stability of sensitive species. If habitat is already fragmented by land use, further stress may further endanger freshwater species.

Various taxa may be sensitive to environmental change, including climate change. Fish are sensitive to temperature, and changes in temperature may shift the ranges of some species, possibly causing local extinctions (Fiske et al., 2005). Changes in water chemistry and limnology may also affect fish. For example, increased temperature reduces dissolved oxygen and increases thermal stratification (Fiske et al., 2005). Some amphibians may experience reproductive issues, such as interference with their life cycles or temperature effects on gender determination (Lind, undated). Climate-related changes in the ranges of pathogens or increases in emerging pathogens may also endanger freshwater species.

Coastal Vulnerability Index (#51)

Definition: The Coastal Vulnerability Index, created by Thieler and Hammar-Klose (2000), is intended to be a measure of the relative vulnerability of U.S. coastal areas to the physical changes caused by relative sea-level rise (RSLR) (Thieler and Hammar-Klose, 2000).

Measurement/Calculation: The CVI at a particular location is calculated based on the values of six variables at that location: geomorphology, coastal slope, rate of RSLR, shoreline erosion and accretion rates, mean tidal range, and mean wave height (Thieler and Hammar-Klose, 2000). Each location on the coastline is assigned a risk value between 1 (low risk) and 6 (high risk) for each data variable. The CVI is then calculated as the square root of the product of the ranked variables divided by the total number of variables: CVI = [(a*b*c*d*e*f*)/6)]^1/2. Thus, a higher value of the CVI indicates a higher vulnerability of coast at that location. The data for each of the six variables used to map this indicator were collected at various frequencies.

The CVI changes based on changes in the following variables (see Thieler and Hammar-Klose, 2000):

- Geomorphology, which is a measure of the relative erodibility of different landforms. Landforms may be of the following types, listed in order of increasing vulnerability to erosion or increasing value of CVI: rocky, cliffed coasts, fiords, or fiards; medium cliffs or indented coasts; low cliffs, glacial drifts, or alluvial plains; cobble beaches, estuaries, or lagoons; barrier beaches, sand beaches, salt marshes, mud flats, deltas, mangroves, or coral reefs. For instance, the value of the CVI is relatively higher along the Louisiana coast due to its lower-lying beaches and marshy areas with shallow slopes that are more prone to erosion.

- Coastal slope (percentage), which is a measure of the relative risk of inundation and of the rate of shoreline retreat. Shallower slopes are more vulnerable as they retreat faster than steeper ones, and will result in a higher value of the CVI. The lower and upper bounds for the coastal slope are <0.025% and >0.2% for the Atlantic Coast, <0.022% and >0.115% for the Gulf Coast, and <0.6% and >1.9% for the Pacific Coast.

- Rate of RSLR (mm/year), which is the change in mean water elevation at the coast. Higher rates of RSLR, resulting in a higher value of the CVI, cause loss of land and destruction of the coastal ecosystem. The lower and upper bounds for RSLR are <1.8 mm/yr and >3.16 mm/yr for the Atlantic Coast, <1.8 mm/yr and >3.4 mm/yr for the Gulf Coast, and <-1.21 mm/yr and >1.36 mm/yr for the Pacific Coast. In contrast, the value of CVI is relatively lower along the Eastern Gulf of Mexico coast mostly due to lower rates of RLSR.

- Shoreline erosion and accretion rates (m/year), which is the rate at which the shoreline changes due to erosion or sediment deposition. Positive accretion rates (resulting in lower values of the CVI) lead to more stable shorelines that are less vulnerable to erosion, while positive erosion rates (resulting in higher values of the CVI) lead to loss of coastal land. The lower and upper bounds for shoreline erosion or accretion rates are <–2.0 m/yr (erosion) and >2.0 (accretion) for all U.S. coasts.

- Mean tidal range (m), which is the average distance between high tide and low tide. Coastal areas that have higher tidal ranges (resulting in lower CVI values) are less vulnerable to sea-level rise (Kirwan and Guntenspergen, 2010). The lower and upper bounds for mean tidal range are <1.0 m and >6.0 for all U.S. coasts.

- Mean wave height (m), which is a measure of the energy of the wave. A higher energy wave (resulting in higher values of CVI) has a greater tendency to mobilize sediments along the coasts, thereby increasing erosion. The lower and upper bounds for mean wave height are <0.55 m and >1.25 for the Atlantic Coast and the Gulf Coast, and <1.1 and >2.60 for the Pacific Coast.

Impacts of global climate change and other stressors: The CVI is, as noted above, a direct measure of the vulnerability of coastal ecosystems to RSLR induced by climate change, and it also captures a change in the ecological condition of the coastal area with respect to previous conditions (e.g., lower sea-levels). RSLR, exacerbated by long-term temperature increases, is expected to increase flooding duration as well as salinity stress caused by saltwater intrusion (Mendelssohn and Morris, 2000, as cited in Day et al., 2005). These factors, in turn, will lead to increased RSLR, destroying coastal wetlands, which may not be able to accrete upwards at the same rate (Day et al., 2005).

Erosion Rate (#348)

Definition: Erosion rate is a measure of the rate of long-term soil loss due to erosion. Land use patterns, such the use of land for agricultural purposes or deforestation, can also cause erosion (Yang et al., 2003). Soil erosion is a major non-point pollution source of surface water (Yang et al., 2003). Erosion from runoff events may cause higher levels of nutrients, dissolved organic carbon, and sediment loads in surface water sources (Murdoch et al., 2000). The Erosion Rate indicator can, thus, be used to assess differences in the potential vulnerability of surface water sources as a result of erosion effects.

Measurement/Calculation: The Erosion Rate can be estimated using Yang et al.'s (2003) Revised Universal Soil Loss Equation (RUSLE). This estimate is based on four independent variables: rainfall erosivity, soil erodibility, topography, and vegetation. This indicator only takes into account soil erosion caused by rainfall and flowing water, and for a grid cell with coordinates (i, j) it can be calculated as follows (Yang et al., 2003):

A (i, j) = R (I, j) x LS (i, j) x K (i, j) x C (i, j) x P (i, j)

where R = average rainfall erosivity factor,

LS = average topographical parameter,

K = average soil erodibility factor,

C = average land cover and management factor,

P = average conservation practice factor.

These variables affect the Erosion Rate in the following manner:

- Average topographical parameter is a measure of the slope length and steepness. Erosion Rate increases with steeper slopes and greater slope length.

- Soil erodibility is the average long-term erosive tendency of rainfall and runoff. This, in turn, depends on the texture, proportion of organic matter, soil structure, and permeability. Erosion rate increases with greater erodibility.

- Rainfall erosivity represents the erosive force caused by rainfall and runoff. This, in turn, is dependent on the annual precipitation. Greater rainfall erosivity causes a higher rate of soil erosion.

- Average land cover and management factor is a measure of land use and is calculated as the average soil-loss ratio weighted by the distribution of annual rainfall.

- Average conservation practice factor is a measure of practices that control erosion. For RUSLE, P is assigned a value of 0.5 for agricultural land and 0.8 for mixed agricultural and forest land. Erosion rate decreases with active conservation practices.

Impacts of global climate change and other stressors: Increased precipitation and greater storm intensities induced by global climate change may result in increased transport of sediment, leading to higher erosion rates.

Groundwater Reliance (#125)

Definition: Groundwater Reliance is a measure of the dependence of a community on available groundwater resources. It is defined as the share of total annual withdrawals from groundwater. This indicator is particularly important as a measure of vulnerability in those regions that depend primarily on groundwater for drinking water, irrigation, and industrial and commercial purposes, because surface water supplies may be limited, contaminated, or expensive to use (Hurd et al., 1998).

Measurement/Calculation: This indicator is calculated as the ratio of withdrawals from groundwater to total annual withdrawals from groundwater and surface water (Hurd et al., 1998). The data used to map this indicator were collected every five years.

Impacts of global climate change and other stressors: Long-term changes in the hydrologic cycle, specifically groundwater recharge and surface flows, may make regions with higher groundwater reliance more vulnerable to water shortages. In addition, urbanization may have a significant impact on groundwater availability and stream baseflow. Increased impervious surface area may intercept rainfall that would normally recharge aquifers. The intercepted rainfall may be directed into storm drains and carried to streams, urban lakes, or estuaries (Klein, 1979; Simmons and Reynolds, 1982).

Herbicide Concentrations in Streams (#367) and Insecticide Concentrations in Streams (#369)

Definition: These indicators are defined as the average concentrations of herbicides and insecticides, respectively, in US streams. Pesticides are of acknowledged concern for human health as well as the health of aquatic organisms. Their ingestion may lead to a number of health concerns, including kidney problems, reproductive problems, and cancer. These compounds have been studied primarily in laboratory animals, although some information is based on epidemiological data. Pesticides are a primary drinking water quality indicator, with Maximum Contaminant Levels (MCLs) in place for 24 pesticides, mostly in the µg/L range.

Measurement/Calculation: This indicator is calculated as the average concentration of herbicides (herbicides, herbicide degradates, and fungicides) or insecticides (insecticides, insecticide degradates, and acaricides) for all sampling sites and all sampling events. The data used to map this indicator were collected at various frequencies depending on purpose and collection site.

Impacts of global climate change and other stressors: Environmental changes that may affect the concentrations of pesticides in streams include alterations to the hydrologic cycle (Noyes et al., 2009). Lower precipitation in the summer may lower streamflow and reduce dilution, leading to higher concentrations, although higher temperatures may offset this by increasing pesticide degradation (Bloomfield et al., 2006). If winter precipitation increases, dilution will tend to increase as well. Climate change may also alter how water moves over the land. For example, increased precipitation, or more extreme wet events, may increase overland flow because the capacity of the soil to infiltrate water will be exceeded. Intense summer storms may promote increased runoff if the antecedent conditions are dry because the soil will be more hydrophobic (Boxall et al., 2009). These effects may promote a greater input of suspended solids into streams, increasing the loading of particle associated pesticides. Climate-induced changes to pest migration or ranges may prompt changes in pesticide usage, which may be reflected in inputs to surface water (Chen and McCarl, 2001). Bloomfield et al. (2006) note, however, that direct climate change effects would be difficult to predict, and that secondary effects from land use changes associated with climate change may be more important as controls on inputs of pesticides to surface water.

Herbicides in Groundwater (#373) and Insecticides in Groundwater (#374)

Definition: These indicators are defined as the average concentrations of herbicides and insecticides, respectively, in shallow groundwater. Because groundwater can contribute herbicides and pesticides to streams, concentrations of these compounds in groundwater need to be considered in evaluations of surface waters and aquatic ecosystems. The presence of these toxics provides an indication of potential contributions of these chemicals to streams. As described in the previous entry, they are also a primary drinking water concern, and EPA has set MCLs for 24 of these compounds.

Measurement/Calculation: This indicator is calculated as the average concentration of herbicides (herbicides, herbicide degradates, and fungicides) or insecticides (insecticides, insecticide degradates, and acaricides) for all sampling sites and all sampling events. The data used to map this indicator were collected at various frequencies depending on purpose and collection site.

Impacts of global climate change and other stressors: Changes in precipitation brought on by global climate change may affect groundwater herbicide and insecticide concentrations. Greater winter precipitation would promote the movement of these substances through the soil towards the water table, and large storms in particular may rapidly transport them into groundwater. In addition, during drier summers, less biodegradation occurs in the unsaturated zone, leaving greater amounts of pesticides available to be transported to groundwater. Finally, herbicide and insecticide use may increase if climate change leads to increased prevalence of pests and weeds.

Instream Use/ Total Streamflow (#351)

Definition: A primary consideration for healthy aquatic ecosystems is having adequate water to maintain fish and wildlife habitat, and competing demands for water can be a significant stressor to these ecosystems (Meyer et al., 1999). This indicator describes the competition by expressing instream water needs for fish and wildlife as a percentage of total available streamflow.

Measurement/Calculation: The ratio of instream use to total streamflow can be calculated using three variables: total groundwater withdrawals, mean annual runoff, and groundwater recharge. Groundwater overdraft values can be calculated based on the definition in the WRC (1978) report: Groundwater Recharge – Groundwater Withdrawals. Instream use can be calculated based on the definition in the WRC (1978) report: Streamflow * 0.6. Streamflow is assumed to be equal to runoff. This indicator is then calculated using the formula described in WRC (1978): Instream use / (Streamflow – Groundwater overdraft). The data for these variables were collected at various frequencies: data on groundwater withdrawals were collected every 5 years, data on mean annual runoff were collected as a one-time effort in 1975, and groundwater recharge data were collected as a one-time effort between 1951 and 1980.

Impacts of global climate change and other stressors: Changes in water withdrawals due to population change can decrease the streamflow available for instream use. Alterations in the hydrologic cycle due to climate change might also decrease streamflow in some areas. This would cause the instream use/total streamflow ratio to increase. A WRC (1978) report notes that a ratio > 100 (based on 1975 data) indicates that withdrawals of water are having a deleterious effect on the instream environment. DeWalle et al. (2000), however, discuss the scenario of concurrent urbanization and climate change. They note that urbanization can significantly increase mean annual streamflow and may offset reductions in flow caused by climate change. This indicator serves as a good vulnerability indicator because regions with greater competition between instream flow uses and consumptive uses are more vulnerable to decreases in streamflow resulting from climate change.

Macroinvertebrate Index of Biotic Condition (#460)

Definition: The Macroinvertebrate Index of Biotic Condition indicator (#460) is a composite measure of the condition of macroinvertebrates in streams. Assessing the condition of these macroinvertebrate species is a good measure of the overall condition of the aquatic ecosystem as they often serve as the basic food for aquatic vertebrates and are, therefore, essential to aquatic ecosystems with vertebrate species (U.S. EPA, 2010f ; U.S. EPA, 2006; U.S. EPA, 2004).

This indicator allows qualitative measurements of macroinvertebrate condition to be represented as a numerical value. It can be considered a good indicator of relative vulnerability as it compares macroinvertebrate condition at study sites with those at undisturbed reference sites located in similar ecoregions (U.S. EPA, 2006). Furthermore, this indicator may be tracked over time to determine temporal changes in vulnerability relative to a baseline (U.S. EPA, 2010b).

Measurement/Calculation: The Macroinvertebrate Index indicator is represented by the average Macroinvertebrate Index value in a given area. It depends on field observations of six variables: taxonomic richness, taxonomic composition, taxonomic diversity, feeding groups, habits, and pollution tolerance (U.S. EPA, 2006). Each variable is assessed using the benthic macroinvertebrate protocol in which stream samples are collected and the characteristics of macroinvertebrates in them are assessed (U.S. EPA, 2004). Each variable is assigned a score based on field observations and individual scores are summed to obtained the value of the Macroinvertebrate Index, ranging from 0 to 100 (U.S. EPA, 2006). The data used to map this indicator were collected every five years.

The Macroinvertebrate Index changes based on the following variables:

- Taxonomic richness, which is the number of distinct taxa or groups of organisms. A stream with more taxa, which indicates a wider variety of habitats and food requirements, will be less vulnerable to stress.

- Taxonomic composition, which is a measure of the relative abundance of ecologically important organisms to those from other taxonomic groups. For example, a polluted stream will likely have a higher abundance of organisms that are resilient to pollution with lower representation from other taxa and will be more vulnerable to stress.

- Taxonomic diversity, which is a measure of the distribution of organisms in a stream amongst various taxonomic groups. Higher taxonomic diversity represents a healthier stream that is less vulnerable to stress.

- Feeding groups, which is a measure of the diversity of food sources that macroinvertebrates depend on. A more diverse food chain is representative of a more stable aquatic environment that is less vulnerable to stress.

- Habits, which is measure of the characteristics of different organisms and their preferences for different habitats. A stream environment with more diverse habitats (e.g., streambed sediment, rocks, woody tree roots, debris) supports a wider variety of macroinvertebrates and will be less vulnerable to stress.

- Pollution tolerance, which is a measure of the degree of resilience to pollution of macroinvertebrate species in a stream. Highly sensitive organisms will be more vulnerable to contamination in streams, compared to pollution-resistant ones.

Impacts of global climate change and other stressors: The structure and function of macroinvertebrate assemblages is a reflection of their exposure to various stressors over time, as these organisms have long life-cycles over which they change in response to stress (U.S. EPA, 2004). Stable ecosystems are likely to contain a variety of species, some of which are sensitive to environmental conditions. These sensitive taxa are most likely to be subject to local extirpations when exposed to climate-induced changes in temperature or flow conditions. Similarly, these species may not tolerate increases in precipitation or temperature variation, which subsequently increase the frequency of disturbance events.

Macroinvertebrate Observed/Expected (O/E) Ratio of Taxa Loss (#461)

Definition: The Macroinvertebrate Observed/Expected (O/E) Ratio of Taxa Loss indicator (#461; also known as O/E Taxa Loss) is a measure of the biodiversity loss in a stream (U.S. EPA, 2006). The O/E Taxa Loss directly reflects the vulnerability of an ecosystem based on its loss of biodiversity (U.S. EPA, 2006). It also reflects a change in ecological condition relative to undisturbed reference sites (U.S. EPA, 2006).

Measurement/Calculation: This indicator is represented by the ratio of the taxa observed at a site to the ratio of the taxa expected to be present at that site as predicted by a region-specific model (U.S. EPA, 2006). Observed taxa are assessed using the benthic macroinvertebrate protocol in which stream samples are collected and the characteristics of macroinvertebrates present in them are assessed (U.S. EPA, 2004). Expected taxa are predicted by models developed from data collected at undisturbed or least disturbed reference sites within a region, for each of three major U.S. regions – Eastern Highlands, Plains and Lowlands, and the West (U.S. EPA, 2006). O/E Taxa Loss ratios are represented as a percentage of the expected taxa present, and they range from 0% (i.e., none of the expected taxa are present) to greater than 100% (i.e., more taxa than expected are present) (U.S. EPA, 2006). The data used to map this indicator were collected every five years.

Impacts of global climate change and other stressors: Stable ecosystems are likely to contain a variety of species, some of which are sensitive to environmental conditions. These sensitive taxa are most likely to be subject to local extirpations when exposed to climate-induced changes in temperature or flow conditions. Similarly, these species may not tolerate increases in precipitation or temperature variation, which subsequently increase the frequency of disturbance events. A measure of the loss of sensitive species may thus serve as an important indicator of vulnerability to climate change and other stressors.

Meteorological Drought Indices (#165)

Definition: This indicator is defined as the average value of the Palmer Drought Severity Index between 2003 and 2007. Meteorological Drought Indices provide a representation of the intensity of drought episodes brought on by a lack of precipitation (Heim, 2002). For example, the Palmer Drought Severity Index (PDSI) takes into account precipitation and soil moisture data from a water balance model as well as a comparison of meteorological and hydrological drought (Heim, 2002). The PDSI can be used as a proxy for surface moisture conditions and streamflow (Dai et al., 2004). PDSI trends are also linked to climate patterns such as the El Niño-Southern Oscillation (Dai et al., 1998).

Measurement/Calculation: PDSI values can be calculated per the methodology in Karl et al., 1996. Calculated PDSI values can be obtained from NOAA's NCDC Divisional Data for each of 344 climate divisions. The data used to map this indicator were collected monthly.

Impacts of global climate change and other stressors: Because drought is a well recognized stressor for natural and human systems, indicators of the spatial and temporal distribution of drought severity are relevant to vulnerability to additional external stressors. This is particularly true for climate change, as drought is directly linked to changes in meteorology that themselves are likely to be affected by climate change.

Organochlorines in Bed Sediment (#371)

Definition: This indicator is defined as the average concentrations of organochlorines in bed sediments. As part of its National Water Quality Assessment (NAWQA) program, the U.S. Geological Survey has analyzed organochlorines in bed sediment (USGS, 1999). Although they have not been used for decades, organochlorine insecticides linger in sediments, posing a potential threat to humans and aquatic organisms. For example, any increase of organochlorines in shellfish may find its way into the human food chain. As a vulnerability indicator, organochlorines in sediment are deleterious compounds that can cause ecological condition to deviate from what would be expected in an undisturbed system.

Measurement/Calculation: This indicator is calculated as the average concentration of organochlorines in bed sediments for all sampling sites and all sampling events. The data used to map this indicator were collected at various frequencies depending on purpose and collection site. Long et al. (1995) derived critical levels or breakpoints for sediment metals and chemical contaminants such as pesticides, PAHs, and PCBs for estuarine systems. MacDonald et al. (2000) conducted similar work for freshwater systems.

Impacts of global climate change and other stressors: Any environmental factor that disturbs bed sediment or affects its transport may affect the exposure of humans or aquatic organisms to organochlorines. Dredging of rivers and harbors may resuspend sediments, increasing contact with aquatic organisms. More intense storms may also resuspend sediment. On the other hand, climate-related increase of sediment input to larger water bodies may provide some "burial" of contaminated sediments, especially if the new sediment is uncontaminated.

Pesticide Toxicity Index (#364)

Definition: This indicator combines pesticide concentrations for a stream water sample with toxicity estimates to produce a number (the Pesticide Toxicity Index or PTI value) that indicates the sample's relative toxicity to aquatic life. This method, developed by Munn and Gilliom (2001), allows data for multiple pesticides to be linked to the health of an aquatic ecosystem, and it allows streams to be rank ordered by their PTI values (Gilliom et al., 2006). The PTI value for a stream increases as pesticide concentrations increase. It is a suitable vulnerability indicator in that it attempts to estimate the potential damage to an ecosystem's resilience as a result of pesticides.

Measurement/Calculation: The PTI for each sampling event is calculated by summing the toxicity quotients for all pesticides. The toxicity quotient is the measured concentration of a pesticide divided by its toxicity concentration from bioassays (e.g., a Lethal Dose 50 (LD_{50}) or Effective Concentration 50 (EC_{50}) value) for a selected species. For the present study, the toxicity quotient used was an EC_{50} value for each pesticide for the species *Daphnia*.

Impacts of global climate change and other stressors: Concentrations may change due to environmental factors such as urbanization, whereby increased streamflow may decrease concentrations due to greater dilution or produce greater pesticide inputs through increased sediment input. Potential climate-related effects include decreased streamflow, which may increase concentrations through reduced dilution, or increased precipitation, leading to increased streamflow and hence sediment inputs. Conversely, increased temperature may accelerate pesticide degradation, leading to lower concentrations. However Noyes et al. (2009) note that if water temperature increases, pesticides can become more toxic to aquatic organisms. It is not known if this effect would apply to humans. Determining the toxicity of mixtures of pesticides to humans is extremely challenging; exploring toxicity changes as a result of climate change is an important direction for future research.

Precipitation Elasticity of Streamflow (#437)

Definition: The Precipitation Elasticity of Streamflow indicator is designed to assess the sensitivity of streamflow to changes in precipitation patterns. It measures the sensitivity of streamflow to climate change and is useful in assessing the vulnerability of regions where maintaining relatively constant streamflow is critical (Sankarasubramanian et al., 2001).

Measurement/Calculation: The Precipitation Elasticity of Streamflow (E_P) is defined as a change in streamflow caused by a proportional change in precipitation. It can be calculated as follows:

$$E_P\,(P, Q) = \frac{dQ}{dP}\,\frac{P}{Q}$$

where P = precipitation and Q = streamflow.

An indicator value greater than 1 indicates that a large change in precipitation is accompanied by a relatively smaller change in streamflow, and thus, streamflow is elastic or sensitive to precipitation changes. An indicator value of less than 1 indicates that a small change in the precipitation is accompanied by a relatively larger change in the streamflow, and thus streamflow is inelastic or less sensitive to precipitation changes. The data for these variables were collected at various frequencies: data on streamflow were collected annually, and data on precipitation were collected monthly.

Impacts of global climate change and other stressors: Streams do not respond uniformly to increased precipitation due to underlying differences in geology, terrain, and other factors. Precipitation elasticity can be used to predict how increased precipitation brought on by global climate change might affect streams in a given region. Increases in precipitation and storm intensity could result in disproportionately large adverse effects, such as flooding, in areas with high precipitation elasticity. Further, these effects could be enhanced or offset by changes in temperature. Climate change, as well as anticipated increases in urbanization, both contribute to the expected increase in the intensity of storms in some areas, leading to more flooding and severe erosion in flashier stream systems.

Ratio of Reservoir Storage to Mean Annual Runoff (#449)

Definition: The Ratio of Reservoir Storage to Mean Annual Runoff indicator is a measure of the storage capacity of reservoirs relative to runoff within the basin (Graf, 1999). Dams can be used to manage water resources to ensure a reliable supply of water to regions that depend on surface water (Lettenmaier et al., 2008). On the other hand, dams can also alter riparian ecosystems and hydrologic processes, causing unnatural variability in streamflow when water is released, fragmenting aquatic ecosystems, and causing erosion and sedimentation (Graf, 1999). The ability to store a large portion of water from land runoff indicates that a community already has the capacity to harness more surface water if needed and may, therefore, be less vulnerable to changes in hydrologic processes. Arid or semi-arid regions, where water is scarce, tend to have larger reservoirs, some of which may be able store up to three or four times the volume of annual runoff (Graf, 1999). This indicator is a good indicator of the vulnerability of water supply. However, it may have a limited ability to predict the vulnerability of water quality and aquatic ecosystems as dams tend to adversely affect both these variables, while they benefit water supply or availability.

Measurement/Calculation: The Ratio of Reservoir Storage to Mean Annual Runoff is determined by the magnitude of its individual components, reservoir storage capacity and mean annual runoff. The storage capacity of reservoirs in a given region is determined by the size of the dam, and the mean annual runoff is determined largely by precipitation and snowmelt. The data used to map this indicator include runoff data that were collected as a one-time effort between 1951 and 1980, and dam inventory data for which the collection frequency is unknown.

Impacts of global climate change and other stressors: Climate change may introduce increased inter- and intra- annual variation in runoff. Areas with relatively low reservoir storage compared to the availability of runoff may be more vulnerable to intense and prolonged droughts or changes in the seasonal timing of runoff.

Ratio of Snow to Precipitation (#218)

Definition: The Ratio of Snow to Precipitation is the ratio of the amount of snowfall to the amount of total precipitation. It can also be described as the percentage of precipitation falling as snow. As such, a decreasing ratio can indicate either a relative decrease in snowfall or relative increase in rainfall, although annual trends in the Ratio of Snow to Precipitation primarily reflect the former (Huntington et al., 2004).

Measurement/Calculation: The data used to map this indicator were collected annually.

Impacts of global climate change and other stressors: Changes in the Ratio of Snow to Precipitation are driven by temperature variations (Karl et al., 1993). Thus, the ratio will be affected by temperature changes associated with global climate change. Trends in the Ratio of Snow to Precipitation can lead to changes in runoff and streamflow patterns because of the effect on the timing and amount of spring snowmelt (Knowles et al., 2006; Huntington et al., 2004). Because of this, areas with decreasing ratios can be more vulnerable to summer droughts (Feng and Hu, 2007).

Ratio of Water Withdrawals to Annual Streamflow (#219)

Definition: The Ratio of Water Withdrawals to Annual Streamflow indicator is a measure of a region's water demand relative to the potential of the watershed to supply water. This indicator is defined as the share of total annual water withdrawals (from surface water and groundwater) to the unregulated mean annual streamflow (Hurd et al., 1998). Streamflow is important for the sustenance of surface water supply as well as for riparian ecosystems. It is also important for aquifers that are fed by streamflow.

Measurement/Calculation: Unregulated mean annual streamflow is calculated based on drainage area, mean annual precipitation, and mean annual temperature using regional regression models specified by Vogel et al. (1999). The ratio of water withdrawals to annual streamflow can then be calculated using water-use data. The data for these independent variables were collected at various frequencies: mean annual precipitation data were collected monthly, mean daily maximum temperature data were collected monthly, and water-use data were collected every five years.

Impacts of global climate change and other stressors: Regions with higher water demand will withdraw higher amounts of water from streamflow both for immediate use as well as for storage in reservoirs. These regions also rely on institutional management to maintain the critical flow in rivers and streams (Hurd et al., 1998). In the long-term, such regions are likely to be more vulnerable to climate changes that lead to large changes in streamflow. Regions where water demand is a smaller proportion of the unregulated streamflow are likely to be less vulnerable to climate-induced changes in streamflow because there is greater available supply from which to draw without affecting the critical flow (Hurd et al., 1998).

Runoff Variability (#453)

Definition: Runoff Variability is defined as the coefficient of variation of annual runoff. This indicator largely reflects the variation of annual precipitation (Lettenmaier et al., 2008; Maurer et al., 2004). Small or moderate changes in precipitation can lead to larger changes in runoff amounts, increasing runoff variability (Burlando and Rosso, 2002; Karl and Riebsame, 1989). Runoff is also linked to and affected by other factors, such as temperature, evapotranspiration, snowmelt, and soil moisture, and it is a critical component of the annual water balance (Gedney et al., 2006; Maurer et al., 2004; Wolock and McCabe, 1999; Karl and Riebsame, 1989). Furthermore, clearcutting and urbanization also lead to increased runoff.

Measurement/Calculation: Annual runoff can be calculated by aggregating the monthly runoff values for each year. Mean and standard deviation of the annual runoff are calculated, following which the coefficient of variation (i.e., the runoff variability) is calculated by dividing the standard deviation by the mean annual runoff. It is easier to measure runoff than it is to measure other variables in the water balance, such as precipitation and evapotranspiration, thus making it a more reliable indicator (Wolock and McCabe, 1999). The data used to map this indicator were collected every three hours.

Impacts of global climate change and other stressors: Understanding inter-annual variation in runoff is important for future scenarios in which climate change will affect precipitation and temperature, both of which affect runoff (Maurer et al., 2004). The spatial and temporal variability of runoff is also essential for predicting droughts and floods (Maurer et al., 2004).

Stream Habitat Quality (#284)

Definition: The Stream Habitat Quality (#284) indicator is used to assess the condition in and around streams. Physical features such as instream vegetation, sediment, and bank vegetation create diverse riparian habitats that can support many plant and animal species (Heinz Center, 2008). Streams degraded by human use are characterized by decreased streambed stability, increased erosion of stream banks, and loss of instream vegetation. Such streams are marginal habitats for most species (Heinz Center, 2008) and hence may be particularly vulnerable to additional stresses. Stream habitat can be altered quickly due to stochastic events such as major flooding, or slowly over time due to subtle changes in flow regime.

Measurement/Calculation: The Stream Habitat Quality indicator is represented by the Rapid Bioassessment Protocol score, an index that can be used to assess the condition of underwater and bank habitats. The Rapid Bioassessment Protocol is a methodology developed by

EPA to assess habitat conditions based on field observations of ten variables: epifaunal substrate/ available cover, embeddedness (for riffles) or pool substrate characterization (for pools), velocity and depth regimes (for riffles) or pool variability (for pools), sediment deposition, channel flow status, channel alteration, frequency of riffles or bends (for riffles) or channel sinuosity (for pools), bank stability, bank vegetative protection, and riparian vegetated zone width (U.S. EPA, 2004). Each of these variables is observed and assigned a qualitative category and score: Poor (0-5), Marginal (6-10), Sub-optimal (11-15), or Optimal (16-20) (U.S. EPA, 2004). The scores for all the parameters are summed to obtain the Rapid Bioassessment Protocol score for that stream (U.S. EPA, 2004). A higher Rapid Bioassessment Protocol score indicates higher Stream Habitat Quality, while a lower Rapid Bioassessment Protocol score indicates a degraded stream.

Stream Habitat Quality changes based on changes in the following variables (U.S. EPA, 2004):

- Epifaunal substrate or available cover, which measures the relative quantity and variety of natural structures in the stream, such as cobble (riffles), large rocks, fallen trees, logs and branches, and undercut banks, available as refugia, feeding, or sites for spawning and nursery functions of aquatic macrofauna. The abundance of these structures in the stream creates niches for animals and insects, and it allows for a diversity of species to thrive in the same habitat.

- Embeddedness in riffles, which measures the extent to which rocks (gravel, cobble, and boulders) and snags are buried in the silt or sand at the bottom of the stream. Fewer embedded features increase the surface area available to macroinvertebrates and fish for shelter, spawning, and egg incubation. Similarly, pool substrate characterization is a measure of the type and condition of bottom sediment in pools. Firmer sediment, such as gravel, and rooted aquatic vegetation support more organisms.

- Velocity and depth regimes for riffles, which measure the variety of habitats caused by different rates of flow and stream depth, such as slow-deep, slow-shallow, fast-deep, and fast-shallow. The ideal stream habitat will exhibit four patterns which represent the stream’s ability to maintain a stable environment. Pool variability is a measure of the different pool types, such as large-shallow, large-deep, small-shallow, and small-deep. The more diverse the pool types, the greater the diversity of the habitat that can be supported by the stream.

- Sediment deposition, which is a measure of the amount of sediment accumulation in streams. More sediment deposition is indicative of unstable streambeds which are an unfavorable environment for aquatic organisms.

- Channel flow status, which is the extent to which the stream channel is filled with water. Low channel flow may not cover the streambed and vegetation, leaving them exposed and reducing available habitat for organisms. Optimal channel flow covers the streambed, creating more available habitat in which organisms can thrive.

- Channel alteration, which is a measure of the significant changes, typically human-induced, in the shape of the stream channel, such as straightening, deepening, diversions, or conversion to concrete. Altered channels are often degraded and limit the natural habitat available to organisms.

- Frequency of riffles, which is a measure of the number of riffles in a stream. Riffles provide diverse habitats in which many organisms can thrive. Similarly, channel sinuosity in pools is a measure of the degree to which the stream meanders. More sinuous streams allow for diverse natural habitats and can also adapt to fluctuations in water volumes, thereby providing a more stable environment for aquatic organisms.

- Bank condition, which measures the extent to which banks are eroded. Eroded banks indicate moving sediments and unstable stream habitat for aquatic animals and plants.

- Bank vegetative protection, which is a measure of the vegetative cover of the stream bank and near stream areas. Banks with dense plant growth prevent erosion, control nutrients in the stream, and provide shade, thus maintaining a healthier riparian ecosystem. In contrast, banks that are covered with concrete in urban areas or experience high grazing pressure from livestock in agricultural areas prevent vegetative growth along the stream, thereby creating a poorer aquatic environment.

- Riparian vegetated zone width, which is a measure of the extent of the vegetative zone from the edge of the stream bank through to the outer edge of the riparian zone. The riparian vegetated zone buffers the riparian environment from surrounding areas, minimizes runoff, controls erosion, and shades the riparian habitat.

The Stream Habitat Quality indicator allows qualitative measurements of habitat condition to be represented as a numerical value. However, most measurements of independent variables that affect the score are “visual-based”, that is they are dependent on the visual assessment by the field team that will score the study sites for each variable (U.S. EPA, 2004). Despite this, Stream Habitat Quality can be considered a good indicator of relative vulnerability for our purposes as it compares stream conditions at study sites with those at undisturbed reference sites located in similar regions (Heinz Center, 2008; U.S. EPA, 2006). Furthermore, this indicator may be tracked over time to determine temporal changes in relative vulnerability, thus allowing one to assess the impacts of future stressors in relation to present stressors. The data used to map this indicator were collected every five years.

Impacts of global climate change and other stressors: Climate-induced changes in storm intensity, runoff seasonality, average flows, or flow variation could result in disproportionately large negative effects on high quality stream habitats.

Total Use/Total Streamflow (#352)

Definition: This is the second indicator expressing the competition between water needs and water availability in streamflow. According to WRC (1978), the ratio of total use to total streamflow is a measure of the water available for "conflict-free development of offstream uses." It is similar to Indicator #351 (Instream Use/Total Streamflow), except that the numerator includes the needs for both instream and offstream use. It is a good vulnerability indicator because regions that have high offstream needs may be less able to withstand decreases in streamflow that may occur due to climate change.

Measurement/Calculation: The ratio of total use to total streamflow can be calculated using three variables: mean annual runoff, groundwater recharge, and water use. Groundwater overdraft values can be calculated based on the definition in the WRC (1978) report: Groundwater Recharge – Groundwater Withdrawals. Instream use can be calculated based on the definition in the WRC (1978) report: Streamflow * 0.6. Streamflow is assumed to be equal to runoff. This indicator is then calculated using the formula described in WRC (1978): (Instream use + Total Consumptive Use) / (Streamflow – Groundwater overdraft). The data for these variables were collected at various frequencies: mean annual runoff data were collected as a one-time effort from 1951-1980, groundwater recharge data were collected as a one-time effort in 1975, and water-use data were collected every five years.

Impacts of global climate change and other stressors: Meyer et al. (1999) note that climate-induced changes in water availability will occur in a context in which human-induced changes in water demand are also occurring. A reduction in streamflow (e.g., due to changes in climate) or an increase in offstream use (due to greater withdrawals for consumptive use) will increase this ratio. According to WRC (1978), a ratio > 100% indicates a conflict between offstream uses and instream flow needs. As with instream use/total streamflow, total streamflow may be increased by urbanization. This is presumably due to increased impervious area. This may offset any flow reductions due to climate change in areas undergoing population expansion.

Wetland and Freshwater Species at Risk (#326)

Definition: The Wetland and Freshwater Species at Risk is a measure of the level of stress that a watershed is experiencing based on the number of water-dependent species "at risk" (Hurd et al., 1998). The Wetland and Freshwater Species at Risk indicator is defined as the number of aquatic and wetland species that are classified by NatureServe (a non-profit conservation organization that maintains biological inventories for animal and plant species in the U.S.) as vulnerable, imperiled, or critically imperiled. A watershed with a higher value of this indicator might be considered to be more vulnerable than a watershed with the lower value of this indicator.

Assessing the condition of species in a watershed can be a good indication of the health of the watershed. However, the indicator is not necessarily a very strong indicator of the vulnerability of aquatic ecosystems, as it only looks at the absolute number of at-risk species, regardless of the total number of species that occupy that habitat (Hurd et al., 1998). Furthermore, this indicator does not account for the inherent diversity in the watershed; watersheds that historically have more species may be less vulnerable to species loss (Hurd et al., 1998).

Impacts of global climate change and other stressors: Watersheds may be stressed due to changes in the hydrological cycle related to global climate change and encroachment or other disturbances from human activities (Hurd et al., 1998). This may cause populations dependent on affected niches to diminish, and may even lead to extinction of species in some cases (Hurd et al., 1998).

Water Availability: Net Streamflow per Capita (#623)

Definition: Water availability is a measure of the availability of freshwater resources per capita to meet water demand for various human consumptive uses (Hurd et al., 1998). This indicator is defined as the net streamflow per capita.

Measurement/Calculation: This indicator can be calculated as follows:

$$\text{Water Availability} = \frac{(\text{Unregulated annual streamflow} - \text{Annual water withdrawals})}{\text{Population}}$$

This indicator depends on three variables: unregulated mean annual streamflow, water withdrawals, and population living in the watershed. Unregulated mean annual streamflow is calculated based on drainage area, mean annual precipitation, and mean annual temperature using regional regression models specified by Vogel et al. (1999). The data for these variables were collected at various frequencies: mean annual precipitation data were collected monthly, mean daily maximum temperature data were collected monthly, and water-use data were collected every five years.

Impacts of global climate change and other stressors: We might reasonably assume that regions with abundant per capita water availability are less vulnerable to long-term changes in the hydrologic cycle brought on by climate change as well as to population growth, and, conversely, regions with lower per capita water availability are more vulnerable.

5. CHALLENGES PART II: DETERMINING RELATIVE VULNERABILITY

5.1. VULNERABILITY GRADIENTS AND THRESHOLDS

A variety of approaches are available to water quality and natural resource managers who must interpret indicator values and indicator-based vulnerability assessments. These approaches vary depending on the state of available knowledge for a given indicator. In many cases, research suggests that responses of water quality or ecosystem condition to external stressors are linear, meaning that changes in condition (or in indicators of condition) occur over a gradual gradient rather than abruptly. Thus, management decisions can be made based on the value of the indicator along the gradient. In other cases, the response may be non-linear, but the thresholds that distinguish acceptable from unacceptable conditions are not yet fully understood. Given this state of knowledge, management decisions to prevent ecosystem degradation or a risk to human health may be based on the relative value of an indicator along the gradient of known values. For example, managers may act out of an abundance of caution when the value of an indicator increases following a long period of stability, even if the risks associated with inaction are unclear. Managers may also choose to act if an indicator value appears to be significantly different from values in other, more pristine locations.

Another approach is the use of known thresholds to facilitate indicator interpretation by indicating points at which management action is required to prevent adverse impacts to human health and the environment (Kurtz et al., 2001). Vulnerability thresholds *reflect abrupt or large changes in the vulnerability of water quality or aquatic ecosystems.* EPA's Office of Research and Development (ORD) Evaluation Guidelines, which describes key concepts in environmental indicator development, describes the role that thresholds can play in interpreting the values of indicators of ecological condition:

> To facilitate interpretation of indicator results by the user community, threshold values or ranges of values should be proposed that delineate acceptable from unacceptable ecological condition. Justification can be based on documented thresholds, regulatory criteria, historical records, experimental studies, or observed responses at reference sites along a condition gradient. Thresholds may also include safety margins or risk considerations. (U.S. EPA, 2000b).

In this study, we attempted to divide the range of values calculated for appropriate indicators into different classes based on evidence in the literature of abrupt or large changes in vulnerability associated with certain values of the indicator. These functional break points (i.e., objective thresholds that distinguish between acceptable and unacceptable conditions) can be highly useful to decision makers. The literature reviewed for this study, however, most often presented arbitrary cutoffs based on round numbers or frequency distributions. It is not surprising that functional break points do not currently exist for many indicators. Groffman et al. (2006) point out that determining such break points can be challenging due to the non-linear response of many indicators and the multiple factors that can affect the value of functionally relevant indicator break points. For example, natural variation in water chemistry and ecosystem types across the nation leads to spatial variation in critical thresholds for dissolved oxygen (DO). Persistently low DO levels in any one ecosystem can yield a community of flora and fauna that are unaffected by DO levels that would be detrimental to another ecosystem. Blackwater river systems of the Southeastern U.S. illustrate this variation. These systems have high levels of dissolved organic matter that may exceed ecologically relevant thresholds elsewhere in the nation, but locally these are high quality systems that are free from the impoundments that alter other systems in the U.S. (Meyer, 1990).

In some cases, objective break points in non-linear system responses may be characterized through additional research, either through meta-analysis of previous research efforts or through new data collection and analysis. In either case, collection of indicator values associated with a range of ecological responses is required to establish functionally relevant break points. There are several statistical approaches for identifying thresholds in non-linear relationships, including regression tree analysis (Breiman et al., 1984) and two-dimensional Kolmogorov-Smirnov techniques (Garvey et al., 1998). Future research may yield additional insights into how these break points vary spatially (Link, 2005).

In general, we considered three different types of thresholds for the suite of indicators evaluated in this project.

Human health-based thresholds, such as Maximum Contaminant Level Goals (MCLGs) or Health Advisories (HAs), which are set based on scientific studies, can potentially be used as thresholds for water quality indicators. EPA establishes MCLGs for contaminants detected in drinking water based on an extensive review of available data on the health effects of these contaminants.

The MCLG is the maximum concentration of a contaminant in drinking water which has no known or anticipated adverse health effect on the population consuming this water, (U.S. EPA, 2010g; U.S. EPA, 2009b). MCLGs for carcinogens are set to zero, based on any evidence of carcinogenicity, as these effects typically manifest over a lifetime of exposure. MCLGs for non-carcinogens are often based on a Reference Dose (RfD), which is the amount of contaminant that a person can be exposed to daily without experiencing adverse health effects over a lifetime (expressed in units of mg of substance/kg body weight/day). MCLGs are non-enforceable and are based purely on the risk posed by a contaminant to human health (U.S. EPA, 2010c; U.S. EPA, 2009a). The MCLG is, thus, a threshold based on scientific data (as opposed to a Maximum Contaminant Level [MCL] that takes other factors into account[2]).

Similarly, HAs are estimates of acceptable concentrations of drinking water contaminants that are developed by EPA as guidelines to help Federal, State, and local entities better protect their drinking water quality (U.S. EPA, 2009a). Like MCLGs, HAs are not enforceable, but are determined solely based on health effects data, such as exposure and toxicity. Unlike MCLGs, HAs are revised from year to year as new data become available.

Other parameters could also be used to assess the toxicity of a drinking water contaminant (U.S. EPA, 2009c):

- Median Lethal Dose (LD_{50}), which is the oral dose of a contaminant that will cause 50 percent of the population it is administered to die (expressed in mg per kg of body weight);

- Cancer Potency (for carcinogens), which is the concentration of a contaminant in drinking water that poses a risk of cancer equivalent to 1 in 10,000 individuals or 10^{-4};

[2]In contrast MCLs are National Primary Drinking Water Regulations (NPDWRs) established by EPA as legally enforceable standards that can be applied to public water systems to ensure safe drinking water supply to the public (U.S. EPA, 2010c). An MCL is defined as the “highest level of a contaminant that is allowed in drinking water” (U.S. EPA, 2009a). While the MCL is set such that it is as close to the MCLG as possible, it is typically higher than the MCLG as it is determined based not only on health considerations, but also on the sensitivity of analytical techniques available to detect the contaminant as well as on the availability of treatment technologies and the extent to which they can remove the contaminant from drinking water (U.S. EPA, 2009a).

- No Observed Adverse Effect Level (NOAEL), which is the highest dose at which no adverse health effects are observed; and
- Lowest Observed Adverse Effect Level (LOAEL) associated with the RfD, which is the lowest dose at which adverse health effects are observed.

These parameters are considered preliminary or less developed thresholds than an RfD value but could still, potentially, be used as thresholds for drinking water indicators.

Ecological thresholds are central to the ecological theory of "alternate stable states" (Scheffer et al., 2001; May, 1977; Sutherland, 1974; Holling, 1973; Lewontin, 1969), where the biotic and abiotic conditions within an ecosystem can reach multiple equilibria. It is believed that the transition between stable states occurs when a significant perturbation results in the breaching of one or more ecological thresholds. The "ball-in-cup" model is commonly used to illustrate this concept (Beisner et al., 2003). A stable ecosystem can be thought of as a ball that resides at the bottom of a cup. There may be many adjoining cups (i.e., the alternate stable states) that the ball could reside in. Small perturbations may push the ball up the side of the current cup, but the ball will eventually return to the bottom – this steep slope illustrates the concept of *resilience.* If the perturbation is large enough, the ball may be pushed across the lip of the cup (i.e., the ecological threshold) and eventually settle into the bottom of a different cup.

Identifying precise ecological thresholds is widely considered to be a difficult task. Ecosystems can be, and often are, a complex mix of biotic and abiotic elements that are difficult to evaluate. Aside from the complex logistics of examining multiple variables simultaneously over ecologically-relevant timescales, ecosystem evaluations can be complicated by the influence of exogenous factors (e.g., climate, human interference) that introduce uncertainty into observations. Furthermore, it is reasonable to believe that many ecosystems are truly unique, meaning that even if ecological thresholds are well understood, they are not widely applicable for the purposes of understanding vulnerability at broad scales. Finally, in many cases, ecological thresholds are difficult to observe unless breached, and the alternate stable state may not be desirable for social, environmental, or economic reasons. Thus, experiments designed to observe ecological thresholds through artificial induction of an alternate stable state are not commonly implemented.

As the science of alternate stable states advances, it may be possible to define objective thresholds for some of the aquatic ecosystem vulnerability indicators in this study. In the meantime, relative comparisons of indicator values can be made, and the range of values may or may not extend across thresholds that could be used to distinguish between vulnerable and less vulnerable areas.

Sustainability thresholds differentiate between sustainable and unsustainable conditions. In the context of this study, sustainability thresholds are most useful in determining where a water resource may currently be being used unsustainably. The construction of indicators that use sustainability thresholds differs somewhat from other indicators. Instead of directly measuring an environmental condition, they frequently use ratios that attempt to identify whether or not a system is in balance. These ratios may help answer basic questions for a given area, such as “Do groundwater withdrawals exceed groundwater recharge?” Or “Do surface water discharges equal surface water withdrawals?”

The critical value for many ratios centered on these questions is one. For example, for a theoretical indicator evaluating the balance between groundwater withdrawals and groundwater recharge, the indicator values may be calculated as Recharge / Withdrawals. Areas where the value of this ratio is greater than one have more groundwater available than is currently be used and could be considered sustainable (i.e., providing a “safe yield”). These areas could also be considered less vulnerable to additional exposure to stresses that reduce groundwater availability. Conversely, values less than one indicate areas where groundwater withdrawals exceed recharge – a potentially unsustainable condition. These areas would be more vulnerable to further exposure to climate-related stresses that reduce recharge.

We calculated values and produced maps for the 25 indicators described in Section 4.3, and included in Appendix E (displayed using 4-digit Hydrologic Units) and Appendix F (displayed using ecoregions). When available, we applied objective threshold values identified in the literature, as shown in Table 5-1. In these cases, data were divided into two or more categories as specified in the literature. Appendix H includes an evaluation of the extent to which objective functional thresholds may be applicable for each of the mapped indicators. In cases where objective thresholds were not available and visualization of changes in indicator values along a gradual gradient was more appropriate, we produced maps using a continuous grayscale color ramp.

Table 5-1. Indicators with objective thresholds and their vulnerability categories

Indicator	Literature Source	Vulnerability Categories and Thresholds
Instream Use/ Total Streamflow (#351)	Meyer et al., 1999	No thresholds were provided in Meyer et al. (1999). However, the original data source (WRC, 1978) used a threshold of one to indicate regions where water exports are already adversely affecting the instream environment. We displayed this indicator in Appendices E and F with the following categories: <1.00 (sustainable) and ≥1.00 (unsustainable).
Precipitation Elasticity of Streamflow (#437)	Sankarasubramanian et al., 2001	Sankarasubramanian (2001) identified a value of one as a breakpoint between elastic and non-elastic responses in streamflow to precipitation. We displayed this indicator in Appendices E and F with the following categories: <1 (inelastic) and ≥1 (elastic).
Total Use/ Total Streamflow (#352)	Meyer et al., 1999	No thresholds were provided in Meyer et al. (1999). However, the original data source (WRC, 1978) used a threshold of one to indicate a potential conflict between offstream uses and the estimated instream flow needs. We displayed this indicator in Appendices E and F with the following categories: <1.00 (sustainable) and >1.00 (unsustainable).

5.2. MODIFYING AND REFINING INDICATORS TO INCORPORATE THRESHOLDS

A major strength of the approach pursued in this study is the use of readily available data, much of which has been vetted by other researchers, agencies, or institutions. Few indicators, however, directly incorporate objective thresholds. Such thresholds, as noted above, can be highly useful to decision makers, especially when they distinguish between acceptable and unacceptable conditions. In some cases, slight **modification of an indicator definition** can facilitate the identification of objective thresholds. For example, the pesticide indicators (#367, #369, #371, #373, and #374) do not incorporate regulatory or human health thresholds because these indicators are calculated as aggregates of multiple pesticides, some of which are unregulated, and whose health effects are less well understood. As an alternative, a predictive model (Larson et al., 2004) is used to map the average probability of exceeding the human health threshold (maximum contaminant level (MCL)) for atrazine, which is the most commonly used herbicide (Figure 5-1). The predictive modeling approach is currently being expanded by USGS to other pesticides. Because these models are built from variables that may be affected by climate change, they may be particularly well-suited to assessing changes in vulnerability across different scenarios of climate and land-use change.

In addition, new indicators may be developed by **integrating multiple existing data sets**. For example, methylmercury production potential could be a useful indicator of vulnerability of aquatic animals to anthropogenic waste. Currently, there is no existing data source that describes methylmercury potential across the entire U.S. However, a new analysis could be conducted using data for wet soils, temperature, and methylmercury deposition, to assess exposure of aquatic life to this contaminant. Existing data sets could be used for the variables in such an analysis, such as wet soils data from the United States Department of Agriculture-Natural Resources Conservation Service (USDA-NRCS, http://soils.usda.gov/); temperature data from NOAA's NCDC (http://www.ncdc.noaa.gov/oa/ncdc.html); and atmospheric deposition data from the University of Illinois Urbana-Champaign's National Atmospheric Deposition Program (NADP; http://nadp.sws.uiuc.edu/). Development of such aggregate indicators using easily available existing data sets may yield additional useful indicators that are critical for assessing regional vulnerability.

An alternative approach would be to **define ideal water quality and aquatic ecosystem vulnerability indicators**, and then appropriately transform existing data or collect new data to assess vulnerability. Development of indicators that more directly compare the sensitivity and exposure components of vulnerability would facilitate a quantitative comparison of their relative importance. For instance, in an effort to understand the relative importance of temperature and population changes on groundwater availability, water use indicators may have to be scaled relative to water availability or per capita demand. As an example, *groundwater availability per capita* could accommodate adjustments from these diverse influences: precipitation effects on recharge, temperature effects on evaporation, and population effects on demand. The hydrologic component of this evaluation would require a model whose drivers include climate variables, scenarios of whose future values can be developed. Creating primary indicators of ecological function would allow for similar evaluations. Although an approach that defines ideal indicators may yield objective thresholds/breakpoints and clear connections to the three aspects of vulnerability, it is likely that difficulties in collecting all requisite data would limit the number of indicators that could be constructed. However, Figure 5-2 and Figure 5-3 represent examples of two indicators that can be developed using existing data. Figure 5-2 depicts total water use efficiency, a modification of the industrial water use efficiency indicator cited in Hurd et al., 1998. Figure 5-3 depicts total water demand for human uses. Both indicator maps were created using the USGS National Water-Use Dataset to provide a more complete picture of U.S. water use.

The National Environmental Status and Trend (NEST) Indicator Project used another approach to assemble a suite of indicators. The process used in that project included the distillation of many perspectives on water into five **categorical questions** (Table 5-2) that guided the search and development of indicators. All of the questions are addressed to some extent by the indicators mapped during this project, although some key subcategories do not have representative indicators. Some of these indicator classes could be filled by further examination of existing data, but others would require additional data collection efforts. Several published examples of these indicator classes were included in the comprehensive list of indicators first assembled for this project, but were subsequently eliminated based on a lack of data, data gaps, or unreliable quality of the available data sets, or inadequate or incomplete data collection efforts. Data collection or manipulation efforts geared specifically towards informing these indicators, such as those discussed below, might provide the necessary data for creating national-scale maps.

Table 5-2. Vulnerability indicators categorized in the National Environmental Status and Trend (NEST) Framework

Vulnerability indicators from this project categorized according to the question framework from the National Environmental Status and Trend (NEST) Indicator Project.

NEST Question	Example Indicators	Subcategories Not Represented
How much water do we have?	• Meteorological Drought Indices (#165) • Ratio of Snow to Precipitation (S/P) (#218) • Precipitation Elasticity of Streamflow (#437) • Ratio of Reservoir Storage to Mean Annual Runoff (#449) • Runoff Variability (#453)	• Flooding (e.g., Population Susceptible to Flood Risk [#209]) • Groundwater availability (e.g., Groundwater Depletion [#121])
How much water do we use?	• Groundwater Reliance (#125)	• Total water use (e.g., Ratio of Water Use to Safe Yield [#328])
What is the condition of aquatic ecological communities?	• At-Risk Freshwater Plant Communities (#22) • At-Risk Native Freshwater Species (#24) • Stream Habitat Quality (#284) • Wetland and Freshwater Species at Risk (#326) • Macroinvertebrate Index of Biotic Condition (#460) • Macroinvertebrate Observed/Expected Ratio of Taxa Loss (#461)	• Habitat Fragmentation (e.g., InStream Connectivity [#620])
What is the physical and chemical quality of our water?	• Acid Neutralizing Capacity (ANC) (#1)	• Nutrients (e.g., Water Quality Index [#319])
Is the water we have suitable for human use and contact?	• Herbicide Concentrations in Streams (#367) • Insecticide Concentrations in Streams (#369) • Organochlorines in Bed Sediment (#371) • Herbicides in Groundwater (#373) • Insecticides in Groundwater (#374)	• Recreational water quality • Waterborne pathogens (e.g., Waterborne Human Disease Outbreaks [#322])
No clear fit to above questions	• Coastal Vulnerability Index (#51)	

This map displays the probability of predicted concentrations of atrazine, a pesticide, exceeding its regulatory threshold (i.e., its Maximum Contaminant Level or MCL). The resulting map places pollutant concentrations into a human health context.

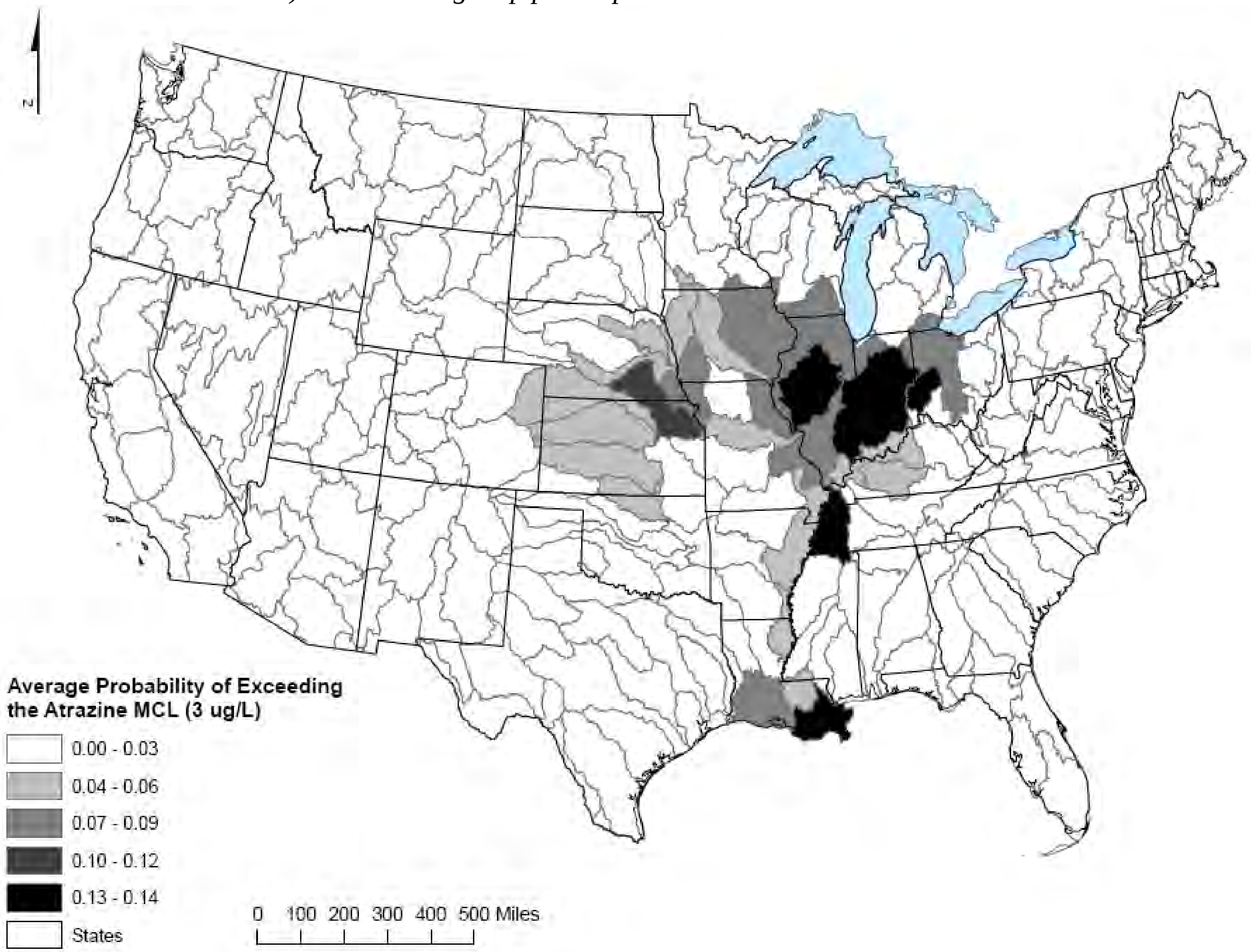


Figure 5-1. Mapping data relative to regulatory thresholds.

This map of 1995 Water Use Efficiency is a refinement of indicator #135. This example demonstrates how minor refinements using existing data sets may result in indicators that more directly assess vulnerability.

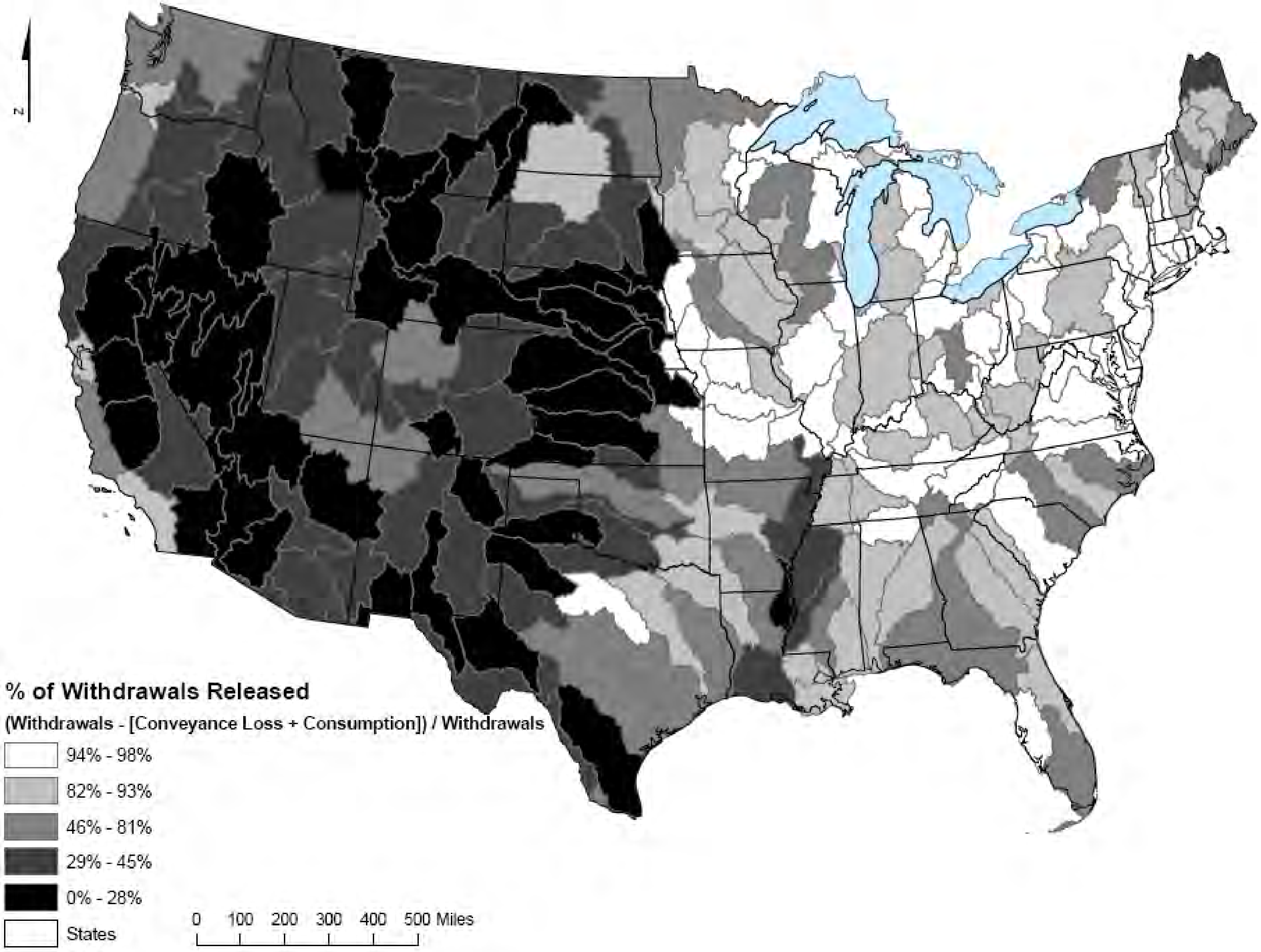


Figure 5-2. Modification of indicator definitions using existing data.

This map of 1995 Water Demand was developed using data sets that were also used to develop indicators #125 and #135. Many of the available data sets used to develop the indicator maps can be used to develop additional indicators of vulnerability.

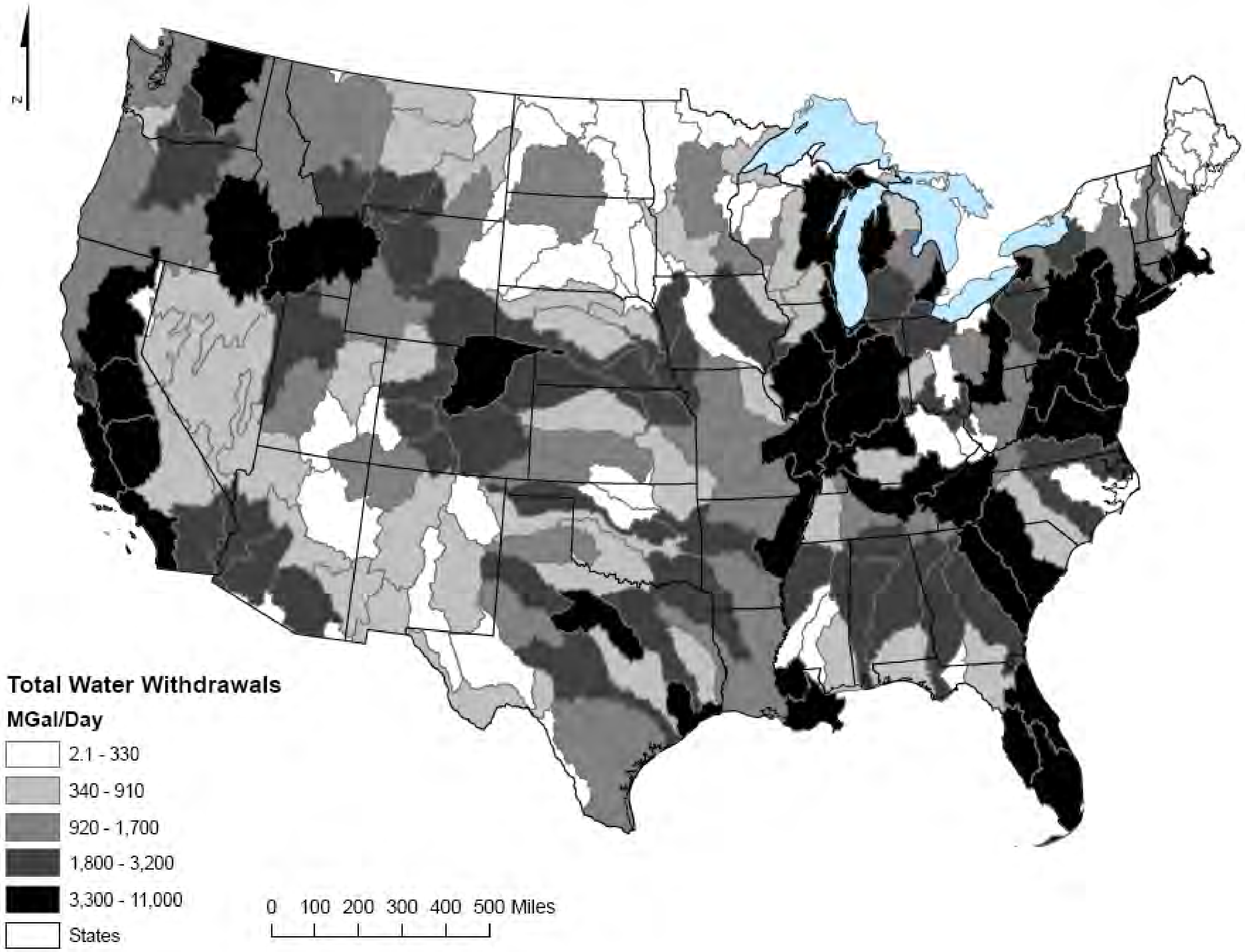


Figure 5-3. Modification of indicator definitions using existing data.

6. CHALLENGES PART III: MAPPING VULNERABILITY

Producing a single map to represent numerical data from disparate sources in an accurate and unbiased manner is a classic cartographic challenge. This challenge is rooted in the fact that "a single map is but one of an indefinitely large number of maps that might be produced…from the same data" (Monmonier, 1996). The choices made with regard to the metrics calculated, the categories used to generalize those metrics, the spatial units used to aggregate localized data, and the symbols used to display map features can all lead to substantially different maps. Furthermore, these choices can be used to emphasize or minimize spatial trends and patterns.

The effort to produce indicator maps for this study was met with these same cartographic challenges. The following sections discuss these challenges in greater detail and provide example maps, using the indicators discussed above, to illustrate how these challenges can affect use of indicators for assessments of vulnerability across the nation.

Mapping the above-described indicators at the national scale requires the compilation of multiple reliable data sets that provide consistent sample density at this scale. In recent years, agencies such as EPA, USGS, and NOAA have invested considerable resources to develop such data sets. These are immensely informative and were used to develop many of the maps contained in this report.

6.1. ASSESSMENT OF INDICATOR DATA AVAILABILITY AND MAPPABILITY AT THE NATIONAL SCALE

We examined the 53 vulnerability indicators (see Table 4-1 and Figure 3-1) for data availability and mappability, in the process identifying existing, available data that could potentially be used for creating national maps for each of these indicators.

6.1.1. Identification of Data Sources for Indicators

We determined data availability for each indicator by re-examining the literature in which the indicator was cited. In most cases, the study that cited the indicator also cited a data set, either one that was collected and assembled during the study itself or a publicly available data set containing data compiled by the authors of the study or by one or more private or public entities. If no specific data set was cited in the original literature, data sets recommended by team members or technical advisors were used. If a data set was not available or could not be recommended, the indicator was marked as having no associated data and was not evaluated for mapping.

Data availability was the most serious limitation in evaluating whether or not we could produce maps for the 53 vulnerability indicators. Of these, only 32 indicators were initially assessed as having adequate data (using data sources identified in the literature) for nationwide mapping. Furthermore, not all of these 32 indicators could be mapped, as the data sources referenced in the literature were not always tailored specifically to the indicator. This was frequently the case with indicators that were identified by one entity and whose data were collected by another entity. In contrast, several indicators identified in USGS' The Quality of Our Nation's Waters report (e.g., Herbicide Concentrations in Streams [#367]; Insecticide in Groundwater [#374]; Organochlorines in Bed Sediment [#371]) are based on NAWQA data that are also collected by USGS.

For indicators that met minimum criteria for availability and for which we identified data sets, nationwide mappability at the level of 4-digit HUC watersheds (as a minimum screening criterion) was assessed simultaneously with data availability. This was because we found that it was not possible to establish mappability without beginning the process of manipulating and mapping the data to determine what obstacles there may be to mapping.

6.1.2. Description of Major Data Sources

The data sets identified for these 53 indicators varied in size, level of detail, quality, and relevance to the indicator. Some data sets were collected specifically with the concerned indicator in mind; in other cases, the indicator was designed with a specific data source in mind. From an initial assessment of data sources, it was evident that major national organizations, such as EPA, USGS, NOAA, and NatureServe, were key players in national-scale data collection efforts for indicators of water quality and aquatic ecosystems. For some indicators, we used data sets produced by other organizations or published in peer-reviewed literature.

A distribution of how often we used data sources from these organizations and other entities for assessing indicator mappability is shown in Table 6-1 (Distribution of data sources). The following 14 indicators (out of 53) had no data available and are, therefore, not included in the 39 indicators in the table: Flood Events (#100), At-Risk Native Marine Species (#27), Freshwater Rivers and Streams with Low Index of Biological Integrity (#116), Harmful Algal Blooms (#127), Invasive Species-Coasts Affected (#145), Invasive Species in Estuaries (#149), Riparian Condition (#231), Status of Animal Communities in Urban and Suburban Streams (#276), Streamflow Variability (#279), Snowmelt Reliance (#361), Salinity Intrusion (#391), Threatened and Endangered Plant Species (#467), Vegetation Indices of Biotic Integrity (#475), and Instream Connectivity (#620). See Appendix C for a complete and more detailed listing of data sources for each of the 39 indicators in Table 6-1.

Table 6-1. Distribution of data source

Acid Neutralizing Capacity (ANC) (#1)	**X** – Wadeable Streams Assessment				
Altered Freshwater Ecosystems (#17)	**X** – National Land Cover data set (NLCD)	**X** – National Hydrography data set (NHD)			**X** – U.S. Fish & Wildlife Service's (USFWS) National Wetlands Inventory (NWI)
At-Risk Freshwater Plant Communities (#22)				**X** – Customized data set	
At-Risk Native Freshwater Species (#24)				**X** – Customized data set	
Coastal Benthic Communities (#462)	**X** – Sampling data in National Coastal Assessment (NCA) database				
Coastal Vulnerability Index – CVI (#51)					**X** – Carbon Dioxide Information Analysis Center's (CDIAC) Coastal Hazards Database
Commercially Important Fish Stocks (#55)			**X** – Annual Commercial Landing Statistics		

Indicator	Data Source Organization				
	EPA	***USGS***	***NOAA***	***NatureServe***	***Other***
Erosion Rate (#348)					**X –** Yang, D. W., S. Kanae, T. Oki, T. Koike, and K. Musiake. 2003. Global Potential Soil Erosion with Reference to Land Use and Climate Changes. Hydrological Processes 17:2913–2928.
Fish and Bottom-Dwelling Animals (#95)	**X** – Wadeable Streams Assessment (WSA)				
Groundwater Depletion (#121)		**X** – National Water-Use Dataset			
Groundwater Reliance (#125)		**X** – National Water-Use data set			
Heat-Related Illnesses Incidence (#392)					**X** – National Center for Health Statistics (NCHS)'s Mortality data
Herbicide Concentrations in Streams (#367)		**X** – NAWQA			
Herbicides in Groundwater (#373)		**X** – NAWQA			
Insecticide Concentrations in Streams (#369)		**X** – NAWQA			
Insecticides in Groundwater (#374)		**X** – NAWQA			
Instream Use/ Total Streamflow (#351)					**X** – Water Resources Council. 1978. The Nation's Water Resources: The Second National Water Assessment, 1975–2000. Volume 2.
Low Flow Sensitivity (#159)		**X** – National Water-Use Dataset			
Macroinvertebrate Index of Biotic Condition (#460)	**X** – Wadeable Streams Assessment				

Indicator	**Data Source Organization**				
	EPA	***USGS***	***NOAA***	***NatureServe***	***Other***
Macroinvertebrate Observed/ Expected (O/E) Ratio of Taxa Loss (#461)	**X** – Wadeable Streams Assessment				
Meteorological Drought Indices (#165)			**X** – Divisional Data on the Palmer Drought Severity Index (PSDI)		
Number of Dry Periods in Grassland/ Shrubland Streams and Rivers (#190)		**X** – Hydro Climatic Data Network (HDCN) & Stream Gauge Data			
Organochlorines in Bed Sediment (#371)		**X** – NAWQA			
Pesticide Toxicity Index (#364)		**X** – NAWQA			**X** – EPA's ECOTOX database
Population Susceptible to Flood Risk (#209)					**X** – FEMA's Q3 Flood Data & ESRI ArcUSA's U.S. Census tract data
Precipitation Elasticity of Streamflow (#437)		**X** – HDCN			**X** – Oregon State University's PRISM Climate Modeling System
Ratio of Reservoir Storage to Mean Annual Runoff (#449)		**X** – Mean Annual Runoff Data			**X** – USACE's National Inventory of Dams (NID)
Ratio of Snow to Total Precipitation (#218)			**X** – Monthly Climate Data		
Ratio of Water Use to Safe Yield (#328)					**X** – Schmitt, C. V., Webster, K. E., Peckenham, J. M., Tolman, A. L., and J. L. McNelly. 2008. Vulnerability of Surface Water Supplies in Maine to the 2001 Drought. *Journal of the New England Water Works Association.* 122(2): 104–116.

Indicator	Data Source Organization				
	EPA	***USGS***	***NOAA***	***NatureServe***	***Other***
Ratio of Water Withdrawals to Annual Streamflow (#219)		**X** – National Water-Use Dataset			**X** – Oregon State University's PRISM Climate Modeling System
Runoff Variability (#453)					**X** – University of Washington's Variable Infiltration Capacity (VIC) Land Surface Data Set
Stream Habitat Quality (#284)	**X** – Wadeable Streams Assessment				
Total Use/ Total Streamflow (#352)					**X** – Water Resources Council. 1978. The Nation's Water Resources: The Second National Water Assessment, 1975–2000. Volume 2.
Water Availability: Net Streamflow per Capita (#623)		**X** – National Water-Use Dataset			**X** – Oregon State University's PRISM Climate Modeling System
Water Clarity Index (#318)	**X** – NCA				
Water Quality Index (#319)	**X** –NCA				
Waterborne Human Disease Outbreaks (#322)					**X** – Centers for Disease Control and Prevention (CDC)'s Waterborne Disease and Outbreak Surveillance System (WBDOSS)
Wetland and Freshwater Species at Risk (#326)				**X** – Customized data set	
Wetland Loss (#325)					**X** –USFWS National Wetlands Inventory (NWI)

As can be seen in Table 6-1, some data sources furnished data for multiple indicators. These major data sources are discussed in greater depth below.

EPA's Wadeable Streams Assessment (WSA)

EPA's WSA was designed to be the first statistically defensible summary of the condition of the nation's streams and small rivers. Chemical, physical, and biological data were collected at 1,392 wadeable perennial stream locations in the coterminous United States. Data were collected by field crews during summer index periods between 2000 and 2004. Sample sites were selected using a probability-based sample design; rules for site selection included weighting based on the 1st- through 5th-order stream size classes and controlled spatial distribution. Due to this sampling system, the sampling effort for the WSA varies across HUC-4 units. Because a probability-based sampling design was used, the WSA data set may have avoided the bias that may occur with ad hoc data sets. However, it is still less than ideal for mapping average conditions in 4-digit HUCs because lakes, reservoirs, and large rivers were not sampled, and because some HUCs had few or no sampling sites.

USGS's National Water-Quality Assessment (NAWQA) Program

USGS's NAWQA Program collects chemical, biological, and physical water quality data. From 1991 to 2001, the NAWQA program collected data from 51 study units (basins) across the United States; after 2001, data collection continued at 42 of the study units. Although the program spanned 10 years, not all 51 sites were sampled every year, but were, instead, broken up into smaller temporal frames (20 study units in 1991; 16 study units in 1994; and 15 study units in 1997).

The NAWQA data warehouse currently contains sampling information from 7,600 surface water sites (including 2,700 reach segments for biological studies) and 8,800 wells. The NAWQA sampling design uses a rotational sampling scheme; therefore, sampling intensity varies year to year at the different sites. In general, about one-third of the study units are intensively investigated at any given time for 3–4 years, followed by low-intensity monitoring. Due to this sampling scheme, the sampling effort for the NAWQA Program varies across HUC-4 units.

USGS' National Water-Use Dataset

USGS's National Water-Use Dataset contains water-use estimates for each county in the United States, the District of Columbia, Puerto Rico, and the U.S. Virgin Islands. USGS publishes reports every five years (starting in 1985) that present water-use information aggregated at the county, state, and national levels. USGS study chiefs from each state are responsible for collecting and analyzing information, as well as making estimates of missing data and preparing documentation of data sources and methods used to collect those data. The study chiefs are also responsible for determining the most reliable sources of information available for estimating water use for each state. Because of this, data sources and quality may vary by location.

NOAA's Monthly Climate Data

NOAA's National Climatic Data Center (NCDC) is the world's largest active archive of weather data. NCDC's Monthly Climate Data Set contains information collected for 18,116 sites across the United States from 1867 to the present. The data set includes an assortment of parameters such as measurements of rain, snow, evaporation, temperature, and degree days. NCDC Monthly Climate data are primarily intended for the study of climate variability and change. NOAA reports that, whenever possible, NCDC observations have been adjusted to account for effects from factors such as instrument changes, station relocations, observer practice changes, and urbanization.

NatureServe Data Set Customized for EPA

NatureServe collects and manages detailed local information on plants, animals, and ecosystems though natural heritage programs and conservation data centers operating in all 50 U.S. states, Canada, Latin America, and the Caribbean. The data sets were originally customized for the Heinz Center for publication in the 2008 *State of the Nation's Ecosystems* report. We obtained updated state-level data on At-Risk Native Freshwater Species (#24) and on At-Risk Freshwater Plant Communities (#22) to produce the maps for these indicators in this study. These data sets were provided in Excel format by NatureServe on July 29, 2009. Data on freshwater species were updated from those presented in the Heinz Center, 2008 report, and included counts of at-risk (GX-G3) and total native freshwater animal species by state for the U.S. Due to incomplete state distribution, the data set did not include giant silkworm moths, royal moths, sphinx moths, or grasshoppers. NatureServe did not update data on plant communities as they determined that plant community data have not changed significantly since the original analysis for the Heinz Center.

6.1.3. Supporting Information Collected for Data Sources

To assess data availability, we isolated information about the underlying data on which the indicators were based. This information is also presented in Appendix C (Data Sources, Supporting Information, and Technical Notes). Information considered when assessing the mappability of data included:

- Data sets used and the organizations or individuals who published or own the data;
- How to obtain the data (download online or contact a specific person/organization) and whether or not payment was necessary to obtain the data set;
- Spatial resolution of data (e.g., state, study sites, HUC level, ecoregion);
- Temporal resolution of data (i.e., frequency of data points and duration of data collection);
- Extent of coverage of data (e.g., national, regional, state, local);
- Type of data source (e.g., survey, census, database, modeled data set);
- Format of data (e.g., Excel tables, GIS shapefiles); and
- Relevant metadata (either as a website or a supporting document).

In many cases, the supporting documentation accompanying the data did not provide all of the abovementioned details. However, the available information has proven useful for prioritizing indicators for further investigation into their mappability.

6.1.4. Lack of Data and Other Unresolved Data Problems

6.1.4.1. *Data Availability Issues*

To streamline the process of determining indicator mappability, we identified issues with data availability and how data were presented as early in the process as possible. We encountered problems both in the effort to locate, access, and download indicator data and in the effort to manipulate, transform, or modify the data so that they could be mapped using GIS software at the appropriate scale. Based on our assessment of data availability, 28 indicators were determined to be non-mappable. Although data sets were available for a few of these indicators, the problems with the data sets could not be reconciled, even with greater time and effort spent on data manipulation and mapping, and, therefore, these indicators were considered non-mappable. These 28 indicators presented one or more of the problems listed in Table 6-2 (Indicators eliminated due to lack of data or unresolved data problems).

Table 6-2. Indicators eliminated due to lack of data or unresolved data problems

Data Availability Problem	Description of the Problem	Example Indicators	Specific Data Availability Problem
Data reported by individual states	Reporting, sampling, and assessment methods vary between states. These indicators are likely to reflect programmatic differences instead of differences in vulnerability.	Fish and Bottom-Dwelling Animals (#95)	The indicator is derived from STORET, a database that relies substantially on self-reported data.
		Waterborne Human Disease Outbreaks (#322)	The WBDOSS datasets relies on voluntary reporting from public health departments within the United States.
		303(d) Impaired Waters[3]	The ATTAINS database relies on data reported by individual states.
Multiple Data Sets	Complete data set could only be obtained by combining more than one data set, as specified in the literature. The effort necessary to combine the data ranges widely.	Population Susceptible to Flood Risk (#209)	This would require combining digital flood data from FEMA (unavailable at time of inquiry) and Census Bureau demographic data.
		Water Quality Index (#319)	Five data sets combined into an index.
		Wetland Loss data (#325)	USFWS' National Wetlands Inventory data are at different scales at different locations.

[3]This indicator was not assigned an indicator ID# because it was not derived from the scientific literature. The indicator was added to incorporate EPA's extensive water quality assessment database.

Data Availability Problem	Description of the Problem	Example Indicators	Specific Data Availability Problem
		Coastal Benthic Communities (#462)	Benthic indices vary by region and it is unclear whether regional indices are comparable.
Data set derived from extensive modeling	Complete data set needed to be recreated with extensive modeling using raw data.	Groundwater Depletion (#121)	Indicator based on a modeled base-flow data set developed by Vogel et al. (1999) and presented in Hurd et al. (1998).
		Low Flow Sensitivity (#159)	Indicator based on a modeled base-flow data set developed by Vogel et al. (1999) and presented in Hurd et al. (1998).
		Streamflow Variability (#279)	Indicator based on a model developed by Vogel et al. (1999) and presented in Hurd et al. (1998).
Data collection in progress	Data are unavailable because collection efforts are in progress.	Instream connectivity (#620)	USGS is currently collecting data on indicator as a part of its National Hydrography Dataset.
Not national in scope	Data are unavailable nationally	Number of Dry Periods in Grassland / Shrubland Streams and Rivers (#190)	The data set identified by the Heinz Center contained an analysis of grassland and shrubland watershed areas for Western ecoregions only.
		Water Clarity Index (#318)	Data are only available for certain U.S. coastal regions.
		Invasive Species – Coasts Affected (#145)	This indicator evaluates invasive species within the context of local land use, a scale that is relatively uncommon. No national datasets have been identified that simultaneously evaluate local land management and the presence of invasive species.
		Ratio of Water Use to Safe Yield (#328)	Data set identified by the source only contains data for the state of Maine.
		Salinity Intrusion (#391)	Data sources cited in the information source, (Poff et al., 2002) are local studies with limited (and non-comparable) data sets. No comprehensive national data sets are known to exist.
Data no longer collected, or are not for	Data are not recent enough (cutoff date varies with the indicator) or are based on future projections.	Waterborne Human Disease Outbreaks (#322)	Data are no longer reported (most recent data are from 2006).

Data Availability Problem	Description of the Problem	Example Indicators	Specific Data Availability Problem
the current time period.		Heat-Related Illnesses Incidence (#392)	Data consist of projections for the years 2020 and 2050.
No data set available.	Data for the indicator are unavailable.	At-Risk Native Marine Species (#27)	The Heinz Center (2008) study, which is the source of this indicator, identifies NatureServe as a potential source of information relevant to this indicator, but acknowledges that data availability is limited to a small set of species.
		Flood Event Frequency (#100)	No data source was identified in this study that could be used to map this indicator at a national scale.
		Freshwater Rivers and Streams with Low Index of Biological Integrity (#116)	There are currently no regional or national data bases that assemble this information for a broad range of taxa.
		Harmful Algal Blooms (#127)	Currently, there are no nationwide monitoring or reporting programs for harmful algal events.
		Invasive Species in Estuaries (#149)	Currently, there are no national monitoring programs for invasive species in estuaries and no agreed-upon methods for combining information on the number of species and the area they occupy into a single index.
		Status of Animal Communities in Urban and Suburban Streams (#276)	The Heinz Center (2008) study, which is the source of this indicator, states that currently available data are not adequate for national reporting.
		Riparian Condition Index (#231)	The Heinz Center (2008) study, which is the source of this indicator, identifies four literature sources that outline various ways to create such an index, but acknowledges that no raw data are currently available.
		Snowmelt Reliance (#361)	The information source (IPCC, 2007a) only has theoretical discussion of indicator. No specific data source is cited.
		Threatened & Endangered Plant Species (#467)	This indicator was provided as an example EPA's National Wetland Condition Assessment. This report does not identify a specific data source for this indicator.
		Vegetation Indices of Biotic Integrity (#475)	This indicator was provided as an example EPA's National Wetland Condition Assessment. This report does not identify a specific data source for this indicator.

Data Availability Problem	Description of the Problem	Example Indicators	Specific Data Availability Problem
		Altered Freshwater Ecosystems (percent miles changed) (#17)	A national database with the number of impounded river miles does not exist. Data from three sources need to be integrated, one of which currently does not provide data in electronic form.
		Commercially important fish stocks (#55)	Data for change in fish stock size over time are not currently available. The change in a fish stock size over time would need to be calculated for each area where fish stock data are available.
Duplicate Indicator	Data are available, but the indicator was a duplicate of another indicator.	Fish and Bottom Dwelling Animals (#95)	

This table highlights two challenges to the adoption and use of indicators at a national scale. First, it draws attention to the issue of measurability. In many cases, a measurable indicator requires a substantial effort to calculate the value at a single location. This may be due to the need for prolonged observation periods, complex sampling protocols, or other factors. For example, Vegetation Indices of Biotic Integrity (#475) uses the relationships between anthropogenic disturbances and observations of plant species, plant communities, plant guilds, vegetation structure, etc. to describe wetland condition. Typically, the highest IBI values represent reference standards or least-disturbed ecological conditions. To collect the data required to calculate an IBI, a trained observer must record multiple parameters in the field for each local IBI score. Though the indicator is measurable and highly useful in the locations where data exist, the effort required to collect data for this indicator at a national scale may be prohibitive.

Second, Table 5-2 highlights how data sources that may otherwise be excellent may be problematic for the purposes outlined in this study. We will discuss the issue of self-reported data in further detail as an example. Data sets that rely on individual state reports are problematic for three reasons. First, the monitoring activities and subsequent reporting may be limited by the availability of the state's resources. This can result in data gaps stemming from varying levels of reporting activity across states. Second, state-based assessments that require sampling from a population (e.g., stream assessments) may not rely on statistically rigorous sampling methods, resulting in sampling that may not be representative. Third, assessment methods may vary from state to state. For example, the assessment and classification methods used by states during the development of the 303(d) impaired waters lists vary substantially among states. Together, these inconsistencies in reporting, sampling, and assessment result in maps that may reflect programmatic differences instead of actual differences in vulnerability. For these reasons, indicators based on national data sets that had national coverage but rely on individual entities to voluntarily report data, (e.g., EPA's Storage and Retrieval [STORET] database for water quality data, CDC's Waterborne Disease and Outbreak Surveillance System [WBDOSS], and EPA's Assessment, TMDL Tracking and ImplementatioN System [ATTAINS] database), were not used in the present study.

Figure 6-1 shows a national map that relies on one such national data set, the ATTAINS database. Panel A shows a map that relies on the total stream-miles designated as 303(d) impaired waters. This first map is problematic because it does not account for large differences in assessment rates across states, or for the fact that overall assessment rates are low. According to the EPA ATTAINS database, only 26.4% of the nation's streams and rivers and 42.2% of the nation's lakes and reservoirs have been assessed for impairments, making it difficult to create national-scale indicators. Panel B attempts to account for differences in assessment rates by showing the percentage of assessed stream-miles that are designated as 303(d) impaired waters. Though this second map is an improvement over the first because it normalizes the assessment effort, the programmatic differences still result in areas that may not appear to be vulnerable simply because sampling and assessment methods vary substantially between states.

The following maps display number (panel A) and percent (panel B) of stream-miles designated as 303(d) impaired waters using data from EPA's Assessment, TMDL Tracking and ImplementatioN System (ATTAINS) database.

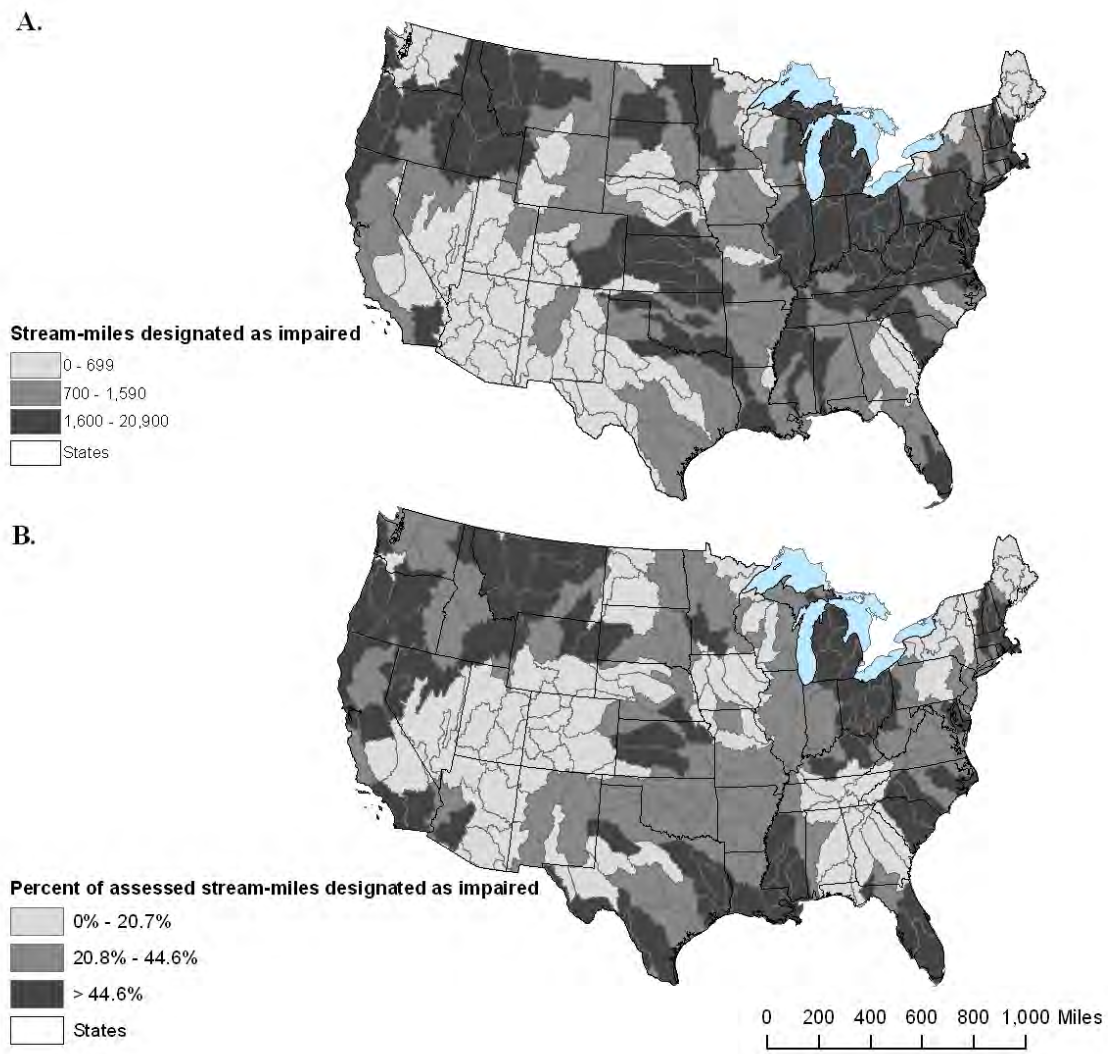


Figure 6-1. Limitations of data sets containing self-reported data.

6.1.4.2. *Data Sets Without National Coverage*

In some cases, the data required to calculate indicator metrics were incomplete in terms of national coverage. Indicators based on a particular ecosystem or land cover type (e.g., grassland or shrubland) may not extend to all parts of the country. For example, few, if any, streams in Eastern ecoregions are grassland or shrubland streams. Other national coverage data gaps stemmed from data availability. For example, although 500 year flood plains can be identified for all parts of the country, GIS-compatible digital flood plain data from FEMA are only available for certain parts of the country where paper maps have been digitized.

Other data gaps were the result of incomplete data collection. For example, for the indicator Commercially Important Fish Stocks (#55), the Heinz Center (2008) study evaluated only about 21% of the commercially important fish landings found in U.S. waters. Similarly, for the indicator Number of Dry Periods in Grassland/Shrubland Streams and Rivers (#190), the data set provided by the Heinz Center contained an analysis of grassland and shrubland watershed areas for Western ecoregions only. Although the reasons for mapping Western ecoregions only are unclear, it is likely that few, if any, sites in Eastern ecoregions satisfied the definition of a "grassland" or "shrubland" watershed used in the 2001 National Land Cover Dataset.

In some cases national coverage was unavailable because data collection efforts are still in progress. For the indicator Wetland Loss (#325), wetlands in 13 states are either unmapped or are recorded only on hardcopy maps. Similarly, data for the indicator Coastal Benthic Communities (#462) (from EPA's National Coastal Assessment [NCA]) and digital flood data for the indicator Population Susceptible to Flood Risk (#209) (from the Federal Emergency Management Administration [FEMA]) were not available at the time of this study for several areas within the U.S.

6.1.4.3. *Non-uniform Spatial Distribution of Data*

In some cases, the national-scale data required to calculate a vulnerability metric are available, however the data are not distributed homogeneously across the country. As a result, varying amounts of data are available within each of the HUC-4 units. This variation can be substantial, and in cases where only few sample points are available within a HUC-4 boundary, individual sites may exert a large influence on the calculated metric value.

The indicator Acid Neutralizing Capacity (#1), for example, is calculated using data from 1,601 stream sites across the country that were sampled as part of EPA's Wadeable Streams Assessment. The number of sites sampled within each of the 204 HUC-4 units varies from 0 to 93, with a median value of 5 sample sites. The calculated vulnerability metrics for HUC-4 units containing the median number of samples (or fewer) are particularly sensitive to measurements at individual sites. A change in the status of a single site from "not at risk" to "at risk" changes the calculated metric (percentage of "at risk" sites) by 20%. This could result in the entire HUC-

4 unit being placed in a different category of vulnerability as a result of a single measurement. A mapping challenge emerges when vulnerability metrics calculated from a small pool of data are mixed with those calculated from a larger pool. It is difficult, and sometimes impossible, to illustrate on a single map where low density would be most likely to result in an erroneous vulnerability classification.

6.1.4.4. *Temporal Gaps*

Many indicators are derived by comparing data contained in two separate data sets, or by comparing data from one data set collected over two distinct time periods. In the first case, it is important to consider the time period in which the data are collected, especially if the information collected may change over time. Temporal gaps between data sets may result in erroneous vulnerability assessments and inaccurate maps. For example, Net Streamflow Availability per Capita (#623) depends on time-sensitive information from a range of data sets. Evaluating streamflow, withdrawals, and population figures from different time periods may provide a different assessment of vulnerability when compared to data collected from the same year. In the second case, indicators based on comparisons to a historical condition are dependent on the existence of historical data. For some indicators considered during the course of this project, this historical information was not available. The Wetland Loss (#325) indicator provides an example of such a case. Information regarding wetland extent is not available at the national scale in a format suitable for mapping with a GIS.

Another issue related to temporal gaps pertains to future data collection. One objective of this project is to identify indicators that can be updated over time to track changes in vulnerability. In cases where data collection and reporting have been discontinued, the indicator no longer meets this key objective. The Waterborne Human Disease Outbreaks (#322) and Runoff Variability (#453) indicators fall into this category. If future data collection efforts are proposed, these indicators may become more useful for national level assessments.

6.1.5. Data Problems that Could be Resolved

Of the 53 indicators that were examined for data availability, twenty-five indicators were mapped. Data sources and supporting information for 32 indicators that had some form of data available that could be examined for mapping are presented in Appendix C (Data Sources, Supporting Information, and Technical Notes).

We identified various types of data gaps in the search for data to represent our vulnerability indicators at the national scale. In some cases, additional assessment of an indicator suggested that there were too many obstacles to nationwide mapping at the present time. Because one rule of thumb for this project was to identify those vulnerability indicators that could be readily mapped, we did not consider indicators that appeared to be mappable but only with extensive data processing efforts. The extent of the data gaps that affected the production of maps differed from one indicator to another, and prohibited production of maps for some indicators. In other cases the problems were minor and maps could be produced (with a few accompanying caveats). The data gaps for this project could typically be placed into one of the three categories shown in Table 6-3.

Table 6-3. Data gaps

Data Availability Problem	Description of the Problem	Example Indicators	Specific Data Availability Problem
Data Sets Without National Coverage	National data collection is incomplete or indicator is location-specific.	Population Susceptible to Flood Risk (#209)	At time of inquiry, GIS-compatible digital flood plain data from FEMA were only available for certain parts of the country.
		Number of Dry Periods in Grassland/Shrubland Streams and Rivers (#190)	Heinz Center data identifies grassland and shrubland watershed areas for Western ecoregions only.
Non-uniform Spatial Distribution of Data	Data are not distributed homogeneously across the country (therefore, number of data points within each HUC varies).	Acid Neutralizing Capacity (#1)	EPA Wadeable Streams Assessment data were collected at 1,601 sites. However, the number of sites within HUC-4 units ranged between 0 and 93 sites.
Temporal Gaps	Lack of historical data (which are needed as a baseline) or time-sensitive data which must be updated frequently.	Wetland Loss (#325)	Historical data on the extent of wetlands is not available.
		Water Availability: Net Streamflow Availability per Capita (#623)	Variables that this indicator depends on (streamflow, water withdrawals, and population) are all time-sensitive. Indicator maps are not useful if recent data are not available.

Mapped indicators typically used nationally recognized data sets or data sets created by national agencies, such as EPA, USGS, and NOAA. While these data sets are comprehensive in nature and cover the entire country, they still have data gaps as well as data quality issues. Nevertheless, the data issues associated with the mapped indicators were either resolved or considered minor enough that a map would still provide useful information for a vulnerability assessment. Minor data issues were carefully documented for the mapped indicators.

6.2. CREATION OF EXAMPLE MAPS

We evaluated for mapping purposes 32 indicators for which national data had been collected. Twenty-five indicators were considered to be mappable (Table 6-4). Six of the remaining indicators were not mapped for this project due to challenges with acquiring data or representing the source data spatially. One of these indicators was mappable, but had substantial gaps in coverage that limited our ability to assess relative vulnerability at a national scale.

Table 6-4. List of mapped vulnerability indicators

Indicator *(See Appendix B for definitions)*	**Literature Source** *(See Appendix A for full citations)*
Acid Neutralizing Capacity (ANC) (#1)	U.S. EPA, 2006b
At-Risk Freshwater Plant Communities (#22) [1]	Heinz Center, 2008
At-Risk Native Freshwater Species (#24)[1]	Heinz Center, 2008
Coastal Vulnerability Index (CVI) (#51) [2]	Day et al., 2005
Erosion Rate (#348)	Murdoch et al., 2000
Groundwater Reliance (#125)	Hurd et al., 1998
Herbicide Concentrations in Streams (#367) [1, 3]	USGS, 1999
Herbicides in Groundwater (#373)[1, 3]	USGS, 1999
Insecticide Concentrations in Streams (#369) [1, 3]	USGS, 1999
Insecticides in Groundwater (#374)[1, 3]	USGS, 1999
Instream Use/Total Streamflow (#351)	Meyer et al., 1999
Macroinvertebrate Index of Biotic Condition (#460)[1]	U.S. EPA, 2006b
Macroinvertebrate Observed/Expected (O/E) Ratio of Taxa Loss (#461)	U.S. EPA, 2006b
Meteorological drought indices (#165)[2]	National Assessment Synthesis Team, 2000a
Organochlorines in Bed Sediment (#371)[1, 3]	USGS, 1999
Pesticide Toxicity Index (#364)	Gilliom et al., 2006
Precipitation Elasticity of Streamflow (#437)	Sankarasubramanian et al., 2001
Ratio of Reservoir Storage to Mean Annual Runoff (#449)[1, 3]	Lettenmaier et al., 2008
Ratio of Snow to Total Precipitation (#218)[2]	Lettenmaier et al., 2008
Ratio of Water Withdrawals to Annual Streamflow (#219)[3]	Hurd et al., 1998
Runoff Variability (#453)	Lettenmaier et al., 2008
Stream Habitat Quality (#284)[1]	Heinz Center, 2008
Total Use / Total Streamflow (#352)	Meyer et al., 1999
Water Availability: Net Streamflow per Capita (#623)[1, 3]	Hurd et al., 1998
Wetland and freshwater species at risk (number of species) (#326)[1]	Hurd et al., 1998

[1]Indicator definition changed based on available data.
[2]Indicator not defined in information source. Definition obtained from primary literature cited in the information source or new definition created based on available data.
[3]Indicator name changed to more appropriately match its definition or the available data.

The software we used for creating the maps for the 25 indicators was ArcMap 9.2 (© 1999–2006 ESRI). For most indicators, data were available either in a GIS format, such as shapefiles, or in tabular form. In some cases, we processed tabular data in Microsoft Excel 2002 or Microsoft Access 2002 prior to importing into ArcMap. In other cases, we manipulated these data and calculated summary statistics directly in ArcMap. We used ArcMap to overlay different data sets, and we ultimately overlaid all data sets with HUC-4 boundaries. The data layer for such boundaries was obtained from the USGS.

For illustrative purposes, we had to choose a spatial unit of analysis. We chose to use USGS hydrologic units at the 4-digit scale here, for three practical reasons. First, USGS hydrologic units provide complete, continuous coverage of the continental U.S., which we established as a requirement of this project. Second, hydrologic units are usually synonymous with watersheds. Using a spatial unit with an inherent link to existing hydrography seems appropriate for a project that is evaluating indicators of vulnerability for drinking water and aquatic ecosystems. HUCs are frequently used by EPA, USGS, and other agencies to monitor water-related phenomena across the country. Finally, 4-digit HUCs were chosen because they balance the need to convey interpretable regional patterns with the objective of providing detailed local information. In other words, in our judgment, they do not over-generalize regional patterns and they do not over-extend the underlying data by providing more local resolution than is warranted. However, we reiterate that the maps we show are to illustrate the various issues we discuss, and we are not advocating any particular spatial aggregation as a matter of best practice. Alternative spatial frameworks or resolutions of course exist, and we discuss the implications for mapping of using such alternatives in more detail in sub-section E (Spatial Aggregation) below.

We aggregated or dis-aggregated the data, depending on their native scale (e.g., state-level data [where there is one data value provided for each state] vs. point data), to obtain a single value of the indicator for each HUC-4 watershed. Using Symbology, we assigned different colors or gray shades to represent the HUC-4 watersheds in different vulnerability categories on each indicator map. The detailed step-by-step methodology for each indicator is documented in Appendix D (Mapping Methodology).

We produced 25 complete example maps by HUC-4 watershed (see Appendix E). In addition, we produced an incomplete map for one indicator for which data suitable for mapping were available for portions of the country. However, substantial gaps in national coverage limit the ability to assess the relative vulnerability of ecosystems to environmental change at a national scale using this indicator. The remaining five indicators were not mapped for this project due to challenges with acquiring data or representing the source data spatially. These issues are discussed in detail below.

The mapped indicators fall into five categories established during the evaluation of the literature (see Section 3), though the indicators we mapped are not distributed evenly across these categories. The categories (with number of indicators mapped shown in parentheses) are: chemical (7); ecological (6); hydrological (8); soil (1); socioeconomic (3).

Assuming that vulnerability can be inferred from metric values that were at the high (or low, depending on the indicator) end of the range of mapped values, regional differences in relative vulnerability were apparent for some of the mapped indicators. For example, the map for the indicator Meteorological Drought Indices (#165) displays high vulnerability in the Western United States, an area that has historically been exposed to prolonged drought. The map also shows high vulnerability for the Southeastern U.S., an area that has experienced a severe drought in recent years.

In some cases, there are no strong regional patterns. For example, the map for Stream Habitat Quality (#284) displays a spatially heterogeneous pattern, with no particular portion of the country strongly distinguished from any other.

Regions for which a single indicator might suggest greater vulnerability may not appear as vulnerable across a full suite of indicators. An examination of the full set of maps by HUC-4 watershed in Appendix E suggests determining overall water quality- and aquatic ecosystem-related vulnerability across all of these dimensions may be complicated. Appendix E also contains detailed descriptions of each of the 25 maps created for the mappable indicators. We return to the issue of combining indicators in more detail in Section 7.

6.3. SPATIAL AGGREGATION

To create a national map illustrating an indicator of vulnerability, it is necessary to aggregate data collected at discrete locations and calculate summary statistics that describe conditions across a larger area. Examples of such statistics may include the mean value of an indicator or the percentage of sites that exceed a threshold value. In many cases, this aggregation process results in a slightly different metric. For example, Acid Neutralizing Capacity is reported in milliequivalents/L at the site scale. However, an aggregate statistic that can be calculated, and is both referred to in EPA’s Wadeable Streams Assessment report and mapped for this report, is the percentage of sites with ANC less than 100 milliequivalents/L. When developing maps using aggregated metrics, it is important for both the producers and consumers of maps to understand how the underlying data and the aggregation methods may affect the validity of objective thresholds and the patterns illustrated in the final map. In the above example, the threshold of 100 milliequivalents/L is a relevant threshold at the scale of an individual site. However, no objective thresholds are defined for the range of aggregated *percentage* values calculated for each HUC. Appendix H includes an evaluation of the effects of aggregation on the validity of theoretical breakpoints for each of the mapped indicators. These issues of aggregation underscore the concept that a single set of data can be used to produce many different maps. The following sections discuss additional factors to be considered when aggregating data.

6.3.1. Local Variation

Measurements at individual sample sites are affected by local factors such as land use, the presence of an industrial facility, an urban center, a protected region (e.g., a National Park), or other features that exist in a heterogeneous landscape. Within a large area (like a HUC-4 unit) that contains a wide variety of these local factors, measurements collected at individual sites may vary substantially. When a group of values within such an area are aggregated into a single value, local variation can be masked. Understanding the degree of local variation is an important component of interpreting vulnerability. For this reason, it may be necessary to simultaneously consider maps that illustrate the vulnerability metric and the variation in raw data values present within each spatial unit.

6.3.2. Extent of Spatial Units (HUC Levels)

Aggregation of individual local measurements into a single metric frequently involves the extrapolation of information. Extrapolation may be appropriate in areas where sampling density is large enough to accurately describe the conditions, and that the extent of the local measurements coincides with the extent of the larger areal unit used to aggregate data. However, extrapolation may also result in the masking of low data density in cases where the extent of the aggregate unit is significantly different from the extent of the underlying data. The producers of maps must be sensitive to the limits of aggregation (and subsequent extrapolation) when choosing a spatial framework to represent a data source comprised of local measurements.

For example purposes, we rely here on 4-digit HUCs to illustrate patterns of vulnerability - we apply it consistently to compare across indicators. For some indicators, however, aggregation of data into this framework may mask low data density. Figure 6-2 illustrates this issue using 3 different scales of HUC units and the same underlying data set. The visual contrast between the top and bottom maps demonstrates how low data density can be masked through aggregation into larger spatial units.

All of the indicators we selected for mapping were chosen based on their ability to provide information on the relative vulnerability of water quality and aquatic ecosystems. As environmental measurements, the data collected and used for each indicator has an inherent level of uncertainty and error associated with it. Selecting a particular unit for presenting information in a set of maps is useful for making comparisons across the set. However, the data collected for the indicators were not available at consistent scales across the set of indicators. The data for most of the indicators was thus altered to present it at a consistent scale. Although manipulating the data changes the accuracy of the information, the manipulations help make the information presented more useful. For the most part, data manipulation required either a scaling up or down of data or transformation of the data from different geographic boundaries.

Data needing to be scaled up included point data. In all cases, the sample data used to calculate metrics for these indicators is not distributed homogeneously. As a result, dissimilar amounts of data are available within the HUC-4 unit boundaries. In cases where there are few sample points within a HUC-4 boundary, individual sites have a greater influence on the metric value that is calculated.

Data presented at the state level needed to be scaled down or transformed to match the HUC-4 geographic boundary. Transforming the data from a state-based representation to a HUC-4 representation requires an assumption that the distribution of the indicator is uniform within each state. Although this assumption is unlikely to be accurate, it allows for area-weighted metrics to be calculated for HUC-4 units that intersect more than one state.

Coastal data presented a unique challenge in mapping. As a watershed geographic unit, HUC-4 has limited or no coverage for coastal and nearshore area data. This makes aggregation for the purposes of reporting at the HUC-4 scale problematic. To address this issue, we developed a special reporting unit for one indicator, the Coastal Vulnerability Index (#51).

Although necessary for creating useful and comparable maps, data manipulations change the quality of the data presented through assumptions about coverage and the representativeness of the data to nearby geographic areas. In most cases, data manipulations are likely to yield greater error and uncertainly than the original data. However, problems associate with data manipulation are likely to be more important for some indicators than others. For example, an indicator based on fine-scale data within a HUC-4 boundary will likely present a more accurate picture of relative regional vulnerability than an indicator based on transformed state-level data.

The following maps display the Stream Habitat Quality (#284) indicator at various scales of HUC units, illustrating how low data density can be masked through aggregation into larger spatial units.

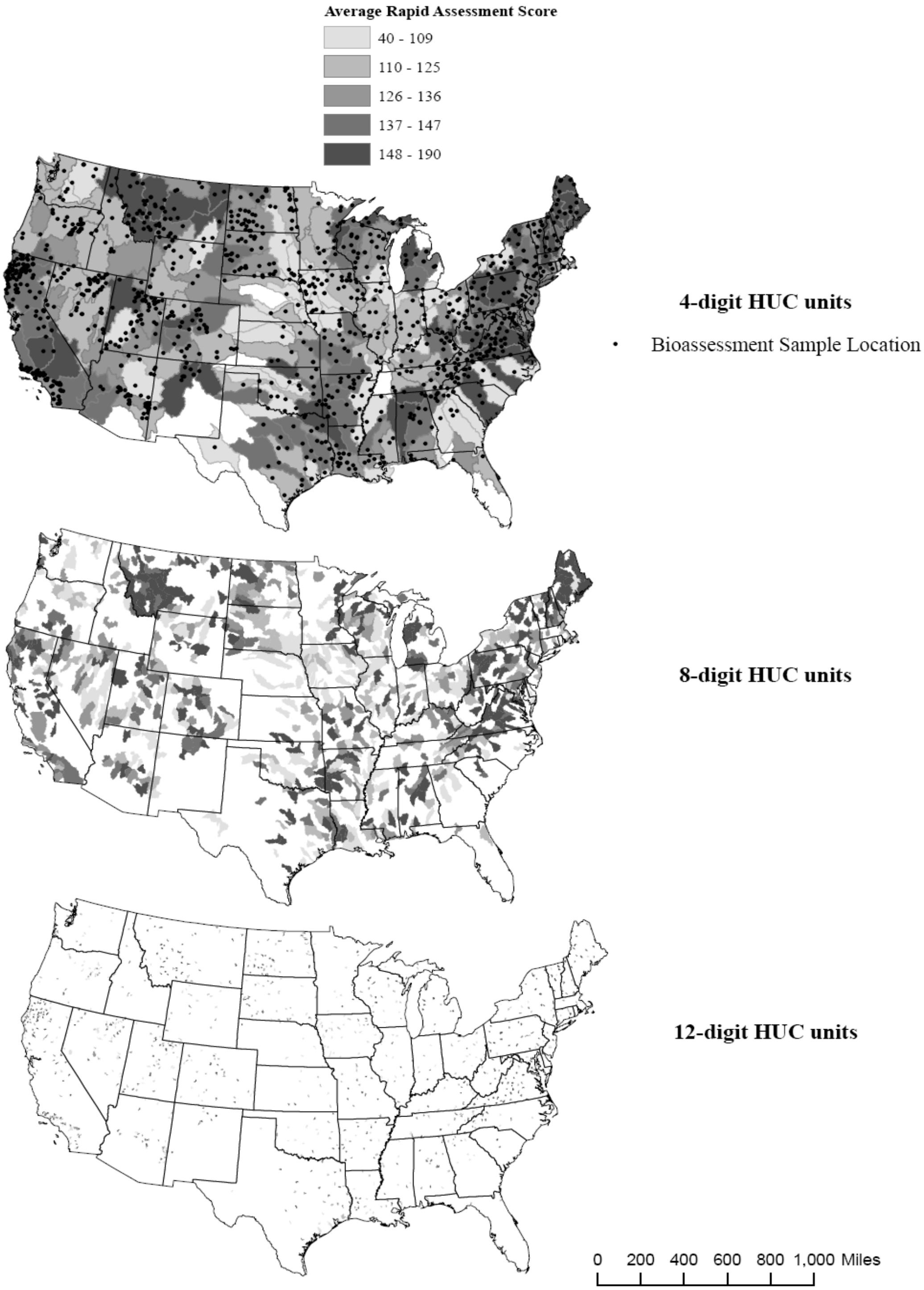


Figure 6-2. Aggregation, precision, coverage, and data density.

6.3.3. Alternate Spatial Frameworks

The selection of the spatial framework used to evaluate geographically-based data can have a significant influence on the graphical display of spatial information and for the assessment and management of resources (Omernik and Griffith, 1991). In some cases, different units of analysis can result in maps that provide difference perceptions using the same set of underlying data. Two spatial frameworks, watersheds and ecoregions, are often associated with ecosystem management. Each of these frameworks has advantages, and the tradeoffs between the two systems reinforce the concept that there is no single best spatial framework for displaying indicators of water quality and aquatic ecosystem condition or vulnerability.

6.3.3.1. *Watersheds (and Hydrologic Units)*

Watersheds are often advocated as the appropriate unit for ecosystem management because they encompass the area of land that influences a connected system of water bodies (Montgomery et al., 1995; U.S. EPA, 1995). To address the practical need for a system of management units that serve as a standardized base for inventorying hydrologic data, the US Geological Survey delineated hydrologic units. These units are commonly identified by their hydrologic unit codes (HUCs) (Seaber et al., 1987). The term “HUC” is often used to describe the hydrologic unit, not just the unit code). HUCs are assigned at several hierarchical spatial scales. The HUC-4 units (n = 204) used in this study have a mean area of 38,542 km^2.

It is noteworthy that many HUCs are true watersheds, while others are combinations of multiple smaller watersheds or segments of a larger watershed. HUCs provide non-overlapping, continuous coverage of a given area, and are typically used in place of true watersheds for mapping environmental data.

6.3.3.2. *Ecoregions*

Ecoregions are alternative spatial units, introduced by Omernik (1987), that are specifically designed to be internally homogeneous with regard to factors that affect water quality, such as vegetation, soils, land forms, and land use. Similar to HUCs, ecoregions are designated at several hierarchical spatial scales. The size of individual ecoregions varies more than individual HUCs. For example, the 87 ecoregions at the Level 3 scale range in size from 649 to 357,000 sq. km.

The shortcoming of ecoregions is that they rarely encompass a single hydrologically connected area, making it difficult to identify the location(s) where cumulative stresses will be felt.

Figure 6-3 illustrates differences resulting from the use of different spatial frameworks. Although the national spatial patterns are similar, there are local differences that may influence vulnerability interpretations. Specifically, differences between the maps are most evident in the western United States – particularly within the Rocky Mountains – and in northern Wisconsin. These differences are reasonable, given the basis for delineating individual areas within each of these frameworks. HUCs, which are based loosely on watershed boundaries, tend to integrate a wider range of physical/topographical characteristics than ecoregions. These local physical characteristics may have a significant influence on the ratio of snow to total precipitation at any one point, resulting in a wide range of values within a HUC. Ecoregions, on the other hand, are specifically intended to describe regions with physical/topographical similarities. Thus, one would expect that ecoregions would contain less within-unit variation for Indicator #218. Maps of the 25 mappable indicators by ecoregion are presented in Appendix F. Appendix F also contains detailed descriptions of each of these maps. From a visual comparison of these maps with the HUC maps presented in Appendix E, it is evident that the choice of similarly sized spatial units (i.e., HUC4 vs. Ecoregion Level 3) has little effect on our results at the national scale.

6.3.3.3. *Coastal Areas*

Coastal areas are worthy of focus in national scale vulnerability assessments because they are of great national importance and pose unique challenges. Coastal areas may be more prone to the effects of climate change, but the limited geographic extent of coastal areas necessitates the use of a different analysis framework. For example, the indicator Coastal Vulnerability Index (#51) uses data available from a USGS database. The data are limited to only coastal and nearshore areas. Although this indicator provides complete coverage of coastal areas, aggregation into HUC-4 units or ecoregions would not provide meaningful results. To address this issue, a set of special reporting units for coastal areas was developed for this indicator. Each unit extends approximately 20 miles inland and includes approximately 150 miles of coastline (Figure 6-4).

6.4. CATEGORICAL AGGREGATION

It is common to symbolize numerical data using chloropleth maps, which use a range of colors that correspond to the underlying data values. Determining how each color is assigned to the range of data values is classic cartographic challenge that applies to most any mapping project, this study included. For numerical data, the methods used to delineate breaks between data classes can affect the spatial patterns conveyed in a map, and the subsequent interpretation of those data. Thus, care must be taken in the development of maps based on numerical data, especially if the resulting spatial patterns may be used to develop policy.

Figure 6-5 illustrates how a single set of data can be used to create alternate maps simply by altering the number of data classes and the breaks used to distinguish between individual data classes.

The following maps display the Ratio of Snow to Precipitation (#218) indicator using 4-digit HUC units and Omernik's (1987) ecoregions, illustrating how the same underlying data appear different when using different spatial frameworks.

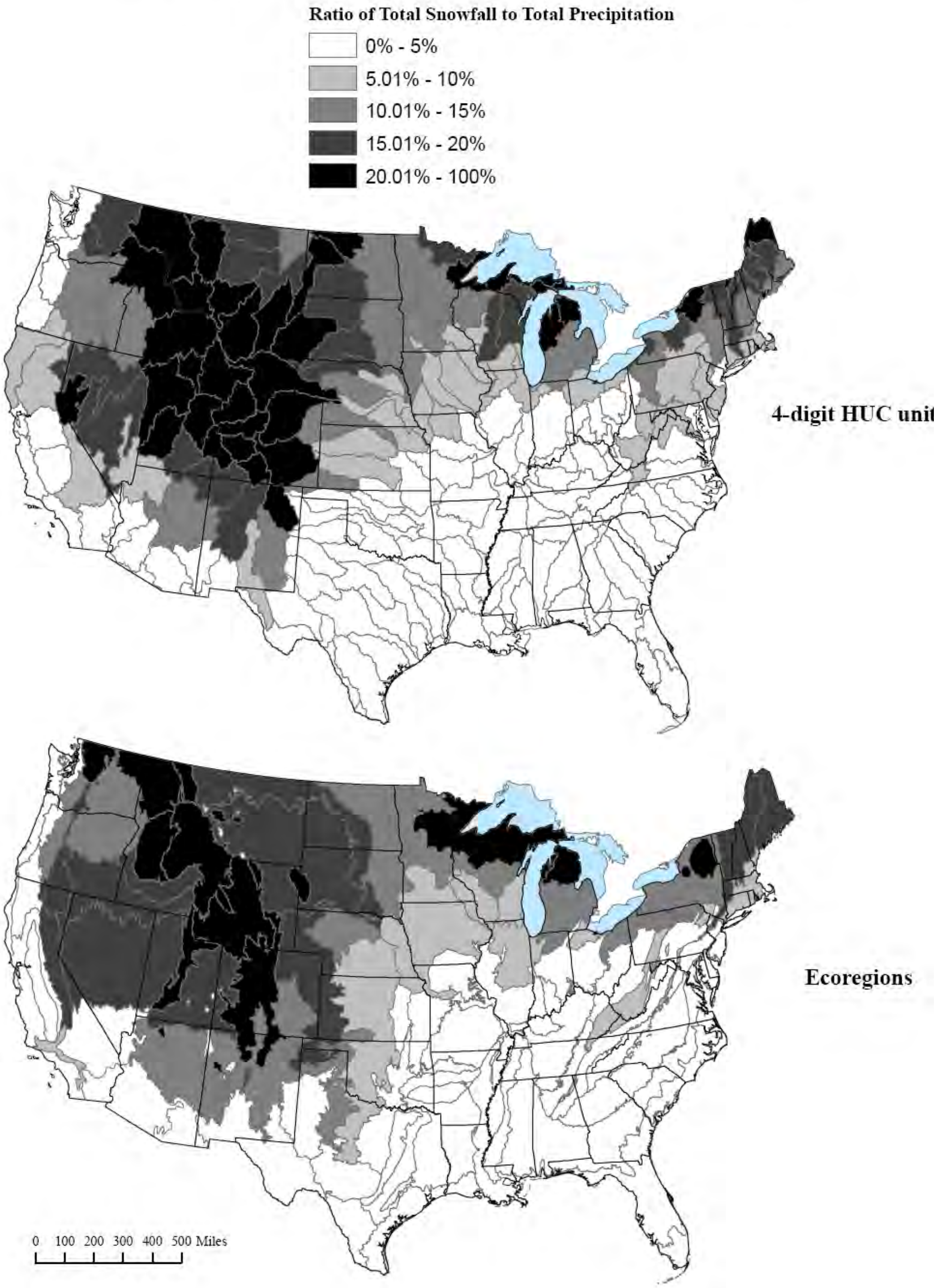


Figure 6-3. Data represented by different spatial frameworks.

The following map displays the Coastal Vulnerability Index (#51), a coastal indicator, for which a set of special reporting units for coastal areas was developed. Each coastal unit extends 20 miles inland and includes approximately 150 miles of coastline.

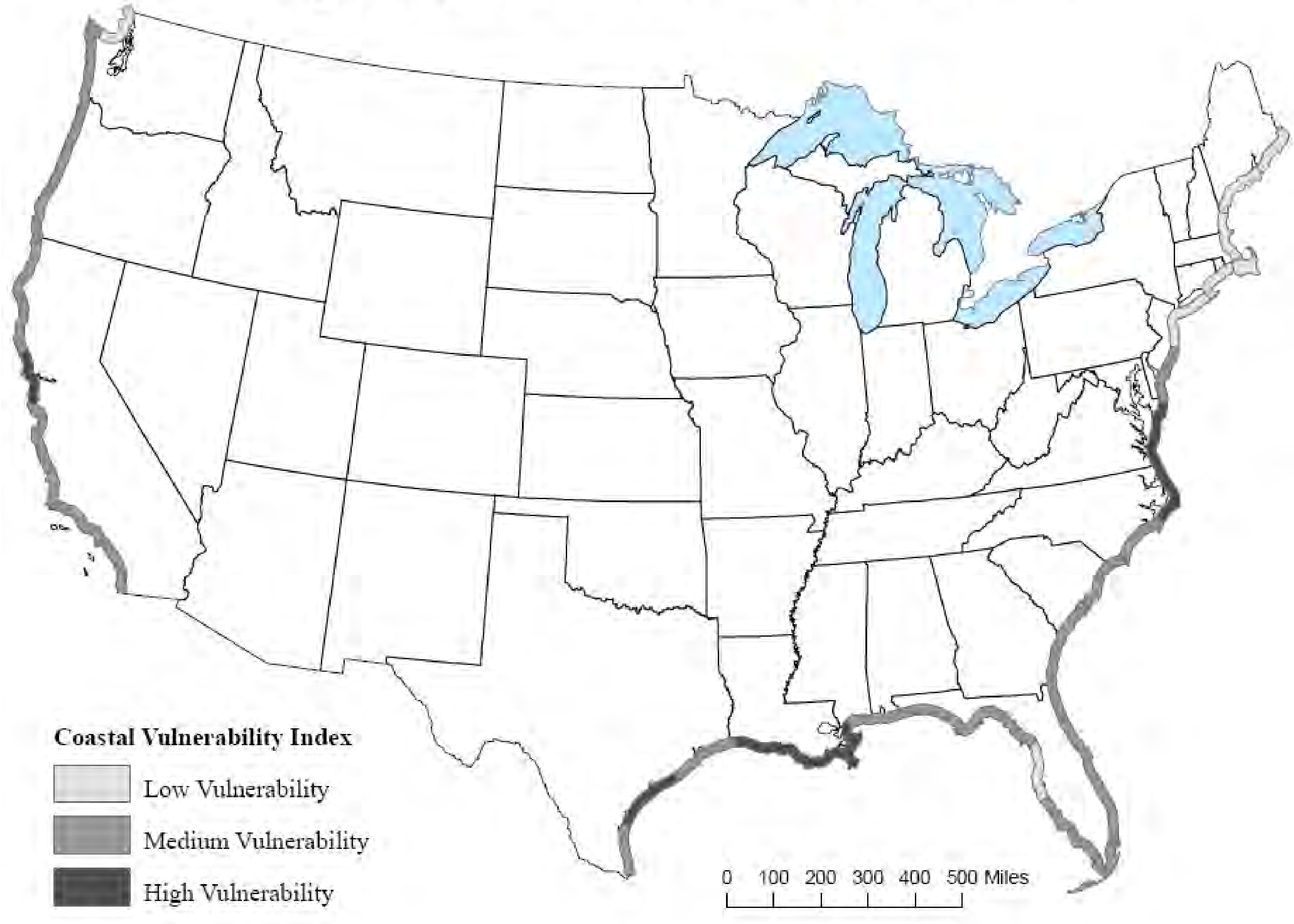


Figure 6-4. Spatial framework for coastal zone indicators.

The following map displays the Stream Habitat Quality (#284) indicator, illustrating how the same underlying data appear different when displayed using three different data breaks (quantiles, equal intervals, and natural breaks or jenks).

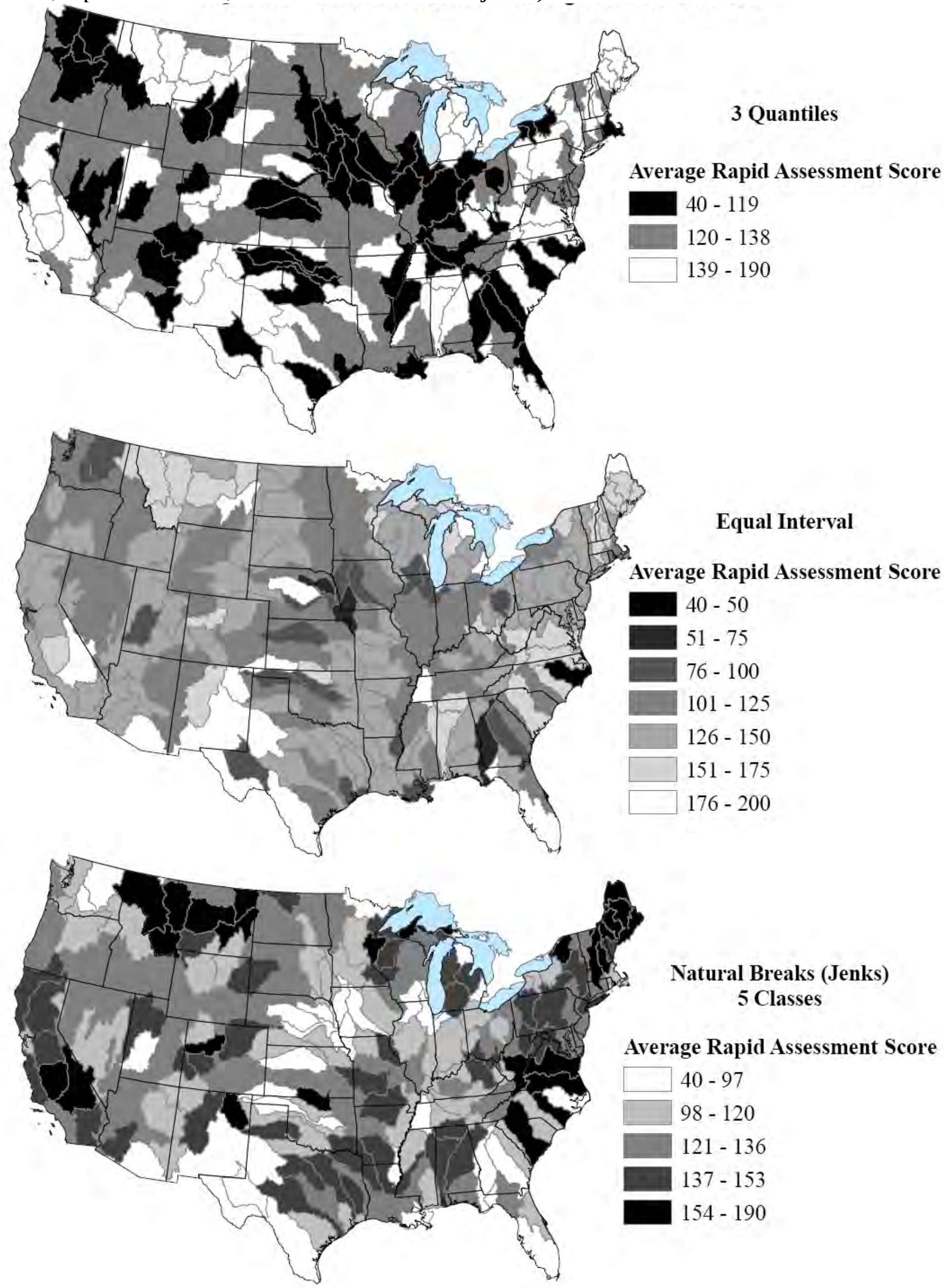


Figure 6-5. Different breaks to distinguish data classes.

7. CHALLENGES PART IV: COMBINING INDICATORS

7.1. COMBINING INDICATORS WITH OTHER DATA

Exposure to future stresses associated with external stressors such as climate and land-use change is likely to vary spatially. Scenarios derived from climate models can be used to map changes in exposure across the plausible range of future changes. A more comprehensive evaluation of future stresses could directly incorporate such scenarios in a vulnerability indicator-based assessment. Figure 7-1 displays an approach for combining indicators identified in this report with other variables. This approach allows the identification of locations that are both vulnerable to stress and are likely to experience additional stress in the future. Four indicators that are related to potential water shortages are presented in the context of simulated changes in temperature and precipitation derived from the IPCC 4th Assessment Report (IPCC, 2007b) and population derived from EPA's Integrated Climate and Land Use Scenarios (ICLUS) project (U.S. EPA, 2009d). Increasing temperature and population and decreasing precipitation all tend to increase the likelihood of water shortages. These plots are examples meant to illustrate how one might go about highlighting regions where we might see a convergence between an already stressed water supply system, a warmer, drier climate, and significant population growth.

While all of the indicators in Figure 7-1 relate to water supply, they deal with different aspects of vulnerability. For example, Precipitation Elasticity of Streamflow (#437) is based only on natural variation in water availability, whereas Groundwater Reliance (#125), Ratio of Withdrawals to Streamflow (#219), and Water Availability: Net Streamflow per Capita (#623) either directly incorporate current rates of water use or infer it through population. These plots illustrate how high water withdrawals in some regions may be unsustainable under a given temperature and precipitation scenario, or how locations that have low water availability per capita might also be places where we expect to see the greatest population increases in the future. In general, under the scenarios used here, current sensitivity and future exposure tend to co-vary, and thus the places that are vulnerable now are likely to become more vulnerable in the future.

The following plots displays values of some example indicators with a sample scenario of temperature and precipitation (based on the B1 greenhouse gas storyline) drawn from the IPCC Summary for Policymakers (IPCC, 2007b) and a population scenario from the Integrated Climate and Land Use Scenarios (ICLUS) project. All variables are scaled as changes over a 100 year period from 2000 to 2100. Each point represents a single HUC-4 and is shaded according to values of the indicator.

A. *Groundwater Reliance (#125) (white, 0–10%; grey, 11–60%; black, 61–100%).*

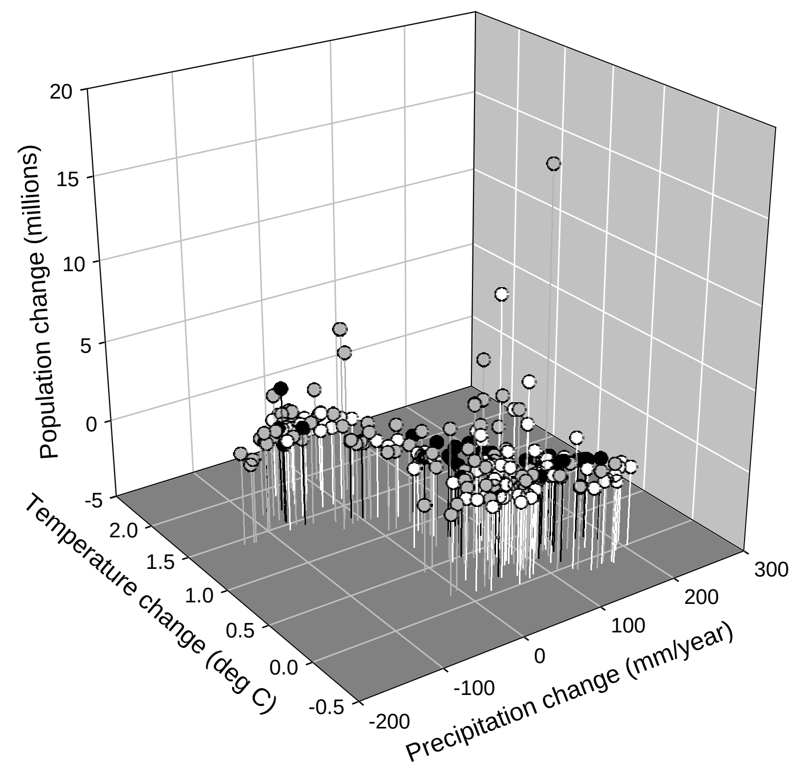
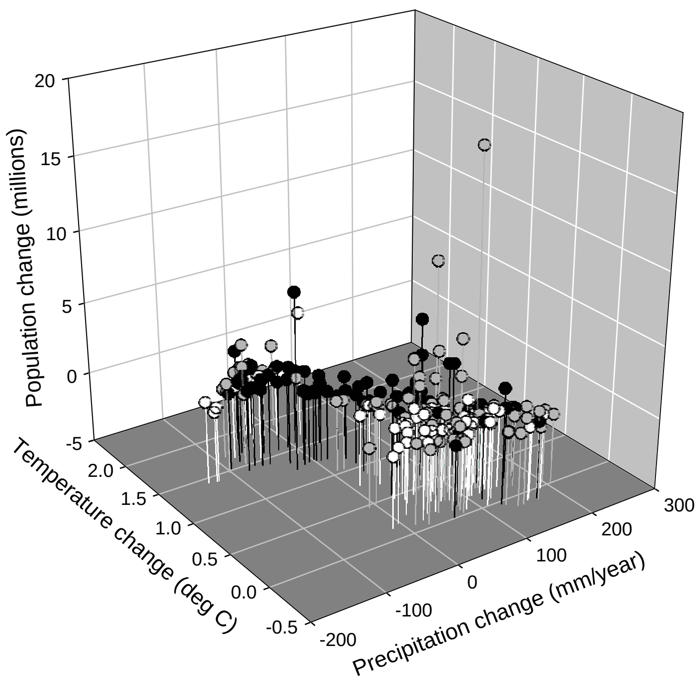
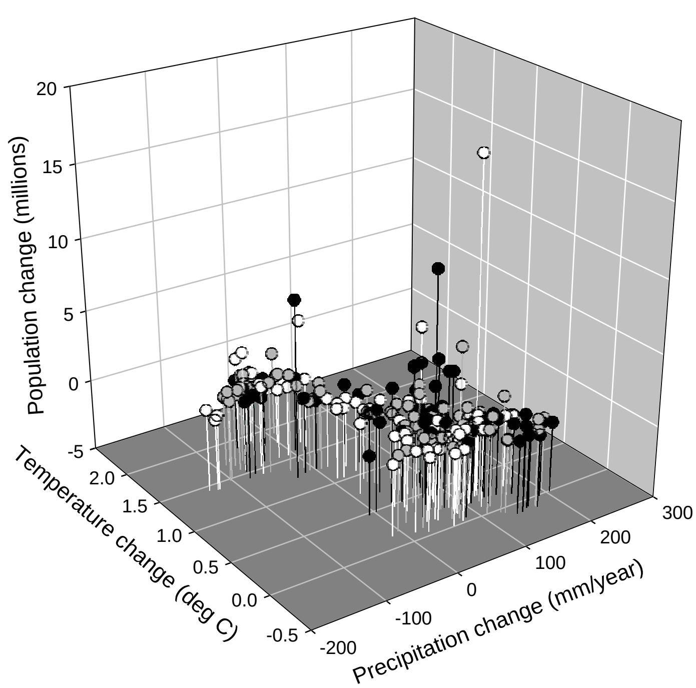


Figure 7-1. Current and future vulnerability to water shortages.

B. *Ratio of Withdrawals to Streamflow (#219) (white, 0–0.11; grey, 0.12–0.75; red, 0.75–59).*



C. *Precipitation Elasticity of Streamflow (#437) (white, 0.43–1.59; grey, 1.60–2.06; black, 2.07–2.96).*



__D.__ Net Streamflow per Capita (#623) (white, 8,493–1,779,536; grey, 888–8,493; black, 0–877).

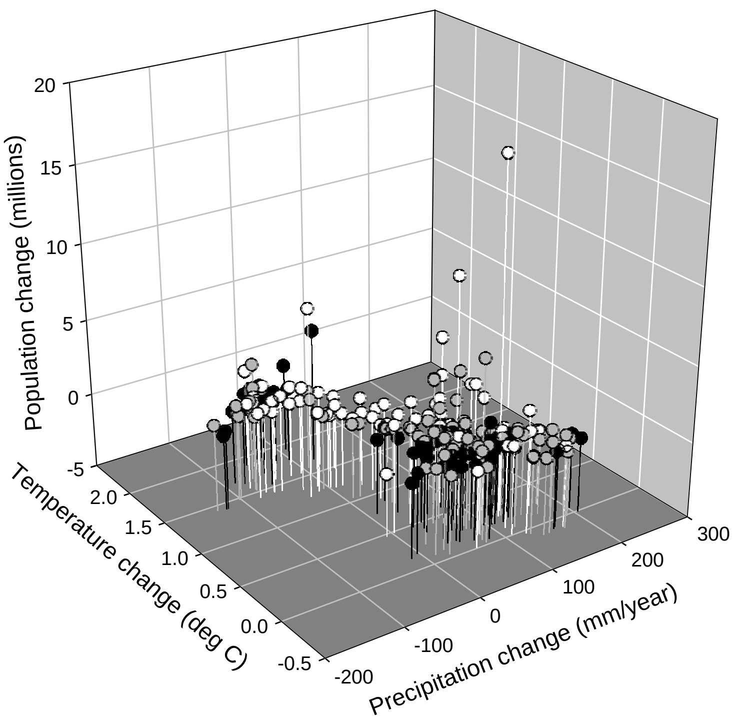


Figure 7-1. Current and future vulnerability to water shortages. (continued)

7.2. COMPOSITES OF VULNERABILITY INDICATORS

Because individual indicators only provide information on limited dimensions of aquatic ecosystem and water quality vulnerability, effective management planning would likely require that these dimensions be integrated into a more holistic perspective on vulnerability. Assuming issues specific to individual indicators can be resolved, there are several possible quantitative methods for integrating multiple indicators.

7.2.1. Creating a Composite Map

Mapped indicators could, potentially, be overlayed into a composite map, such that the averages of all indicator values for each of the HUC units are represented on a single map. This is the approach taken in Hurd et al. (1999). This is challenging, however, for a number of reasons. One major reason is that the distinction between relative and real (i.e., functionally significant) differences in vulnerability, while not necessarily as critical for interpretation of individual indicator maps, is extremely important for the construction of a composite vulnerability map. For example, if the range of values for an indicator only reflect one category of vulnerability (e.g., very high vulnerability), differences in relative vulnerability may be functionally insignificant. If this type of indicator is given equal importance in a composite score to one whose values span a functionally significant range, the composite score will be inaccurate. As a consequence, the vulnerability of individual locations may be under- or over-estimated, depending on the relative frequency of high vulnerability values from these two classes.

Another way to aggregate indicators could be by identifying geographic units where further stresses (including climate change) will cause the most harm across all system dimensions (e.g., see Lin and Morefield, 2011). This can be done as follows:

- Assign numeric scores to the vulnerability categories (e.g., 3 for highest, 2 for medium, and 1 for lowest). Sum the scores across all indicators.
- For each geographic unit, calculate the percentage of indicators that are in the highest vulnerability category.

Once any technical deficiencies and data gaps have been addressed through data collection efforts, construction of a composite vulnerability map should consider the following:

- *The relative importance of system dimensions.* The relative weighting of individual indicators is dependent on management objectives and the degree to which indicators are redundant with one another.
- *Range of indicator values.* Only indicators whose values span functionally significant ranges should be used for a composite vulnerability map. This will lead to a more accurate representation of relative vulnerability.

- *How an integrated vulnerability rating will translate into management or adaptation efforts.* Locations with high integrated vulnerability may either be moderately vulnerable for most attributes, or highly vulnerable for a few attributes. While both of these scenarios point to the need for planning, the specific suite of relevant strategies would differ. Thus, the production of multiple visualization tools may often be a helpful exercise.

7.2.2. Characterizing Vulnerability Profiles

The aim of this type of integrative procedure is to identify commonalities in the types of vulnerabilities among regions. A vulnerability profile for a given location can be defined as the set of values for all the vulnerability indicators. Such an analysis allows watersheds with similar vulnerability profiles to be identified, and might be useful in the transfer of successful management or adaptation strategies from one location to another. Specifically, if a selected watershed is vulnerable in certain ways and in need of an adaptation strategy, other locations with similar vulnerability profiles could be identified. Successful adaptation strategies in those other locations could then be assessed for their applicability in the selected watershed.

Similarities in vulnerability profiles among locations can be summarized numerically through multivariate statistical analyses useful for finding patterns in data, such as Principal Components Analysis (PCA). PCA is used to consolidate the information in a large number of variables into a smaller number of artificial variables (called principal components) that will account for most of the variability in the original variables. The first component extracted in a PCA accounts for the greatest amount of total variance in the original variables, and the second and subsequent components account for progressively less variance.

The principal components (PCs) are described in terms of loadings of the original variables. A PC may be heavily loaded on at least one variable, and usually on more than one. A high loading indicates that the PC is strongly related to that variable (either negatively or positively depending upon the sign of the loading). Variables for which a PC is heavily loaded are correlated with each other, creating clusters of related variables that should be interpretable from a conceptual standpoint. The PCs themselves, however, are uncorrelated with one another. One benefit of conducting a PCA for this study is that reducing the full set of indicators to its principal components helps to avoid overemphasis on system properties that are represented by multiple similar indicators.

As an example, we conducted a PCA on 24 of the 25 mapped indicators (we excluded the Coastal Vulnerability Index (#51) because of its unique spatial units). We normalized indicators with non-normal frequency distributions with log or square root transformations. We inverted the scales of some indicators so that high vulnerability was always represented by high values of the indicator. We used the correlation matrix of these standardized variables for the PCA. When no data were available for an indicator, the HUC was assigned the median value for that indicator. We rotated the PCA (Varimax) and specified a maximum of six principal components – these six cumulatively account for about 57% of the total variance, with 35 % coming from the first three.

Table 7-1 shows the six PCs generated in the PCA analysis. These PCs help demonstrate which types of processes or environmental factors are driving a large part of the variability in the data. PC1 is heavily loaded on indicators related to at-risk species, which are negatively correlated with the ratio of snow to total precipitation (see bolded loadings in Table 7-1). PC2 is correlated with variables indicative of streamflow availability and usage. PC3 represents pesticides in surface water. PC4 is loaded on indicators related to macroinvertebrates and stream habitat quality. For PC5, the most heavily loaded indicator is meteorological drought indices, which is moderately correlated with at-risk freshwater plant communities. Finally, PC6 is loaded on herbicides in groundwater, but not pesticides in groundwater.

Table 7-1. Principal components loadings for the twenty four indicators included in the PCA analysis

Indicator	PC1	PC2	PC3	PC4	PC5	PC6
Acid neutralizing capacity (#1)	0.166	-0.367	-0.231	-0.071	-0.233	-0.265
At-risk freshwater plant communities (#22)	0.401	0.220	-0.007	0.153	0.604	0.090
At-risk native freshwater species (#24)	**0.863**	0.167	0.068	0.051	0.149	0.117
Groundwater reliance (#125)	0.087	0.196	0.242	0.291	-0.033	-0.313
Meteorological drought indices (#165)	0.006	0.182	-0.138	-0.038	**0.771**	0.019
Ratio of snow to total precipitation (#218)	**-0.774**	0.033	-0.167	-0.193	0.120	0.300
Ratio water withdrawal to annual streamflow (#219)	-0.071	**0.873**	-0.089	0.036	0.035	0.056
Stream habitat quality (#284)	0.092	-0.018	0.170	**0.687**	0.196	0.056
Wetland species at risk (#326)	**0.789**	-0.102	0.017	0.026	-0.204	0.200
Erosion rate (#348)	0.387	-0.056	-0.058	-0.076	0.131	0.504
Instream use/total streamflow (#351)	0.132	0.262	0.144	-0.104	0.005	-0.456
Total use/total streamflow (#352)	0.017	**0.753**	0.048	0.126	-0.052	-0.211
Pesticide toxicity index (#364)	0.082	0.009	**0.889**	-0.027	-0.003	-0.041
Herbicide concentrations in streams (#367)	0.078	-0.112	**0.769**	0.111	-0.028	-0.112
Insecticide concentrations in streams (#369)	0.070	0.025	**0.870**	-0.033	-0.020	0.033
Organochlorines in bed sediment (#371)	0.092	0.089	0.515	0.016	-0.358	0.109
Herbicides in groundwater (#373)	0.018	0.212	0.160	-0.009	-0.239	**0.721**
Insecticides in groundwater (#374)	0.191	0.080	0.078	-0.139	-0.537	0.355
Precipitation elasticity of streamflow (#437)	0.628	-0.073	0.156	0.207	0.153	-0.107
Ratio of reservoir storage to mean annual runoff (#449)	-0.117	-0.250	-0.090	0.074	-0.151	0.110
Runoff (variability) (#453)	0.160	0.504	0.036	-0.056	0.256	0.137
Macroinvertebrate index of biotic condition (#460)	0.051	0.074	-0.043	**0.845**	0.007	-0.112
Macroinvertebrate observed/expected (#461)	-0.156	0.030	0.080	**-0.754**	0.066	-0.055
Water availability: streamflow per capita (#623)	-0.150	**0.839**	-0.127	0.002	-0.009	-0.005
Proportion of variability explained	**0.120**	**0.117**	**0.113**	**0.085**	**0.073**	**0.065**

The map in Figure 7-2 is another way of using and displaying the results of the PCA. This map shows the similarity of an example focal watershed (shown in blue) to watersheds across the U.S. We defined the similarity of two watersheds as the weighted Euclidean distance (D_w) among the values of the first six principal components:

$$D_w = \sqrt{\sum_{i=1}^{6} w_i (x_i - y_i)^2}$$

where x_i and y_i are the values of component i for the two watersheds, and w_i is the weight for component i, which is defined as the proportion of the total variance in the entire dataset explained by that component. This approach is similar to the methods used by Tran et al. (2006).

As discussed above, because this kind of analysis and map allows watersheds with similar vulnerability profiles to be identified, it might be useful in the transfer of successful adaptation strategies from one location to another. Specifically, the map could help to identify locations with the most similar multi-dimensional vulnerability profiles to that of a selected focal watershed in need of adaptation strategies. Successful adaptation strategies in those other locations could then be assessed for their applicability at the focal watershed.

While relative similarity could identify the closest matches to the focal watershed, its mean absolute similarity to all other locations would be a measure of its uniqueness. The similarity of all pairwise combinations of watersheds could be cataloged in a vulnerability similarity matrix to expand the applicability of this approach. Such a matrix would include every watershed on the horizontal axis, and these same watersheds on the vertical axis. Each central cell of the matrix would contain a value that documents (according to the formula above) the similarity of the two watersheds defined by that cell. In addition, the vulnerability profile approach could be further refined by applying weights to indicators to account for differences in accuracy or relevance to climate change or other stressors of interest.

The following map displays the results of the PCA conducted on 24 of the 25 mapped indicators. It shows the similarity of the focal HUC watershed (blue) to the remaining 203 watersheds.

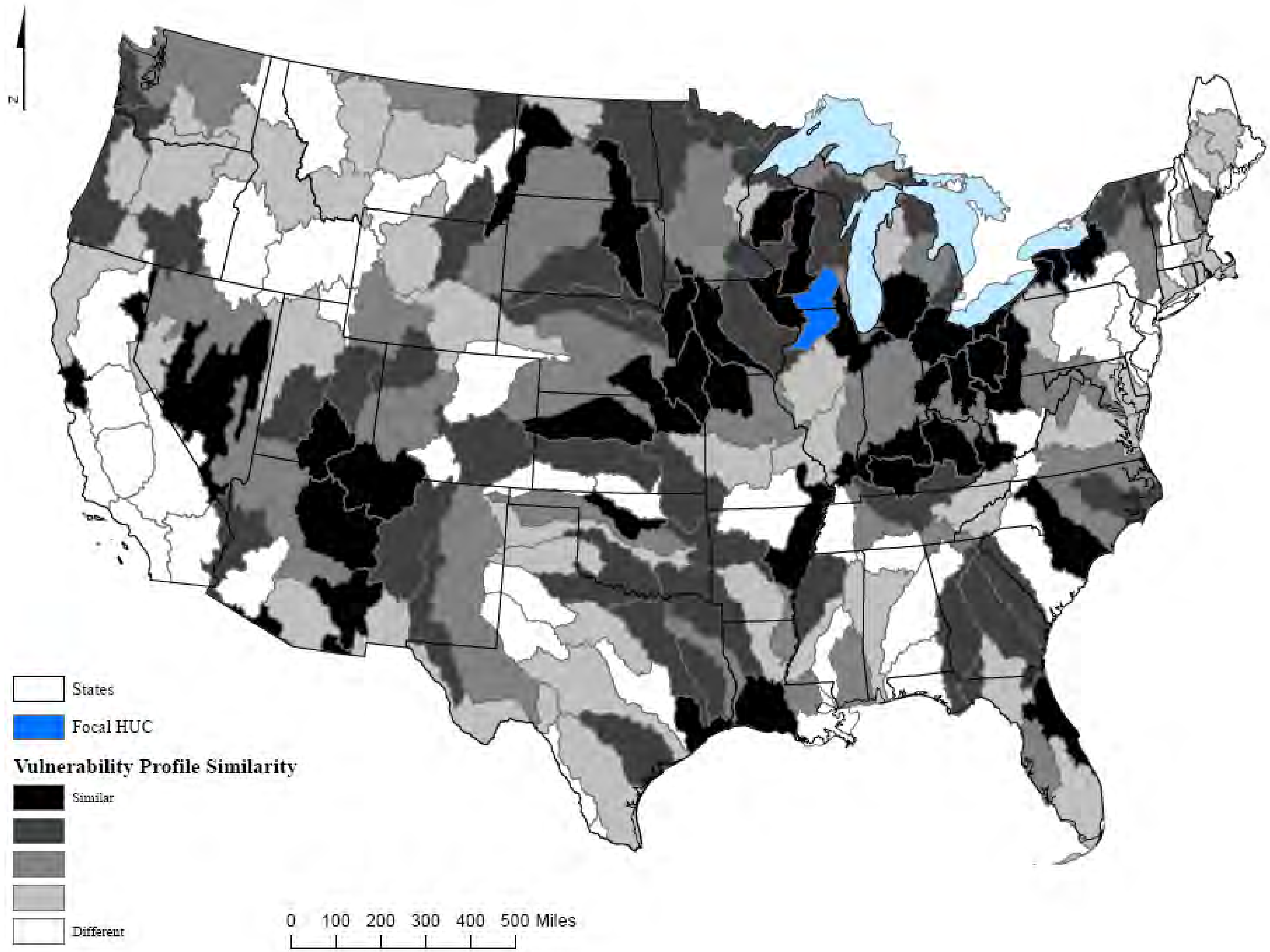


Figure 7-2. Vulnerability profile similarity.

8. SUMMARY AND RECOMMENDATIONS

This report investigates issues, challenges, and lessons associated with identifying, calculating, and mapping indicators of the relative vulnerability of watersheds across the United States to the potential adverse impacts of external stresses such as long-term climate and land-use change. It is our hope that this report will be a useful building block for future work on multi-stressor global change vulnerability assessments.

It is important to clarify here that this report does not attempt any kind of direct evaluation of the potential impacts of climate change or other global change stressors on ecosystems and watersheds. Instead, it deals only with the question of how to estimate the impacts of current stressors. We argue that a systematic evaluation of the impacts of existing stressors is a key input to any comprehensive climate change vulnerability assessment, as the impacts of climate change will be expressed via often complex interaction with such stressors – i.e., through their potential to reduce overall resilience, or increase overall sensitivity, to climate change. This argument is not new, and in fact it has been a staple of writing on climate change impacts, vulnerability, and adaptation, particularly of large assessments like those of the IPCC and U.S. Global Change Research Program. However, to date there has been relatively little exploration of the practical challenges associated with comprehensively assessing how the resilience of ecosystems and human systems in the face of global change may vary as a function of existing stresses and maladaptations.

8.1. SUMMARY OF CHALLENGES

Our approach in this report has two basic elements. First, we have collected, evaluated the quality of, processed, and aggregated a large quantity of data on water quality and aquatic ecosystem indicators across the nation that have been reported on in the ecological, hydrological, and management literature. Second, we have used this set of indicators as a testbed for identifying best practices, challenges, and gaps in ideas, methods, data, and tools for calculating and mapping vulnerability nationally.

Specifically, we compiled a list of 623 indicators of the vulnerability of water quality or aquatic ecosystems that were defined in the literature, focusing our search on expanding the list of indicators rather than reviewing literature for its more general contributions to the body of knowledge on a topic. The indicators compiled relate to drinking water and source water quality, ecosystem structure and function, individual species, and ecosystem services. We explored challenges associated with using these indicators to assess vulnerability of water quality and aquatic ecosystems nationally. These challenges fall into four broad categories:

1. Challenges associated with identifying those indicators that speak specifically to vulnerability, as opposed to those reflecting simply a state or condition;
2. Challenges associated with determining relative vulnerability using indicators, including interpreting gradients of indicator values, and, when possible, establishing important indicator thresholds that reflect abrupt or large changes in the vulnerability of water quality or aquatic ecosystems;
3. Challenges associated with mapping these vulnerability indicators nationally, including data availability and spatial aggregation of the data; or
4. Challenges associated with combining and compositing indicators and developing multi-indicator indices of vulnerability.

Sources of indicator definitions and data used to map the indicators included published research and studies by EPA, other federal agencies, the Heinz Center, the Pew Center, etc. We limited the study to existing indicators and datasets, and for the most part did not attempt to develop new indicators or collect new data. As part of this work, we developed a number of example maps, and we use some of these maps in this report for illustrative purposes. We hope that the lessons we learned while developing strategies for compiling and mapping national-level indicator datasets under this project will be useful for indicator-based vulnerability assessments in general. Here we summarize the main findings of the report, organized according to the four challenges listed above.

8.1.1. Challenges Part I: Indicator Classification

There is on ongoing debate in the literature on the meaning of vulnerability and the elements of which it is composed, particularly in the context of climate change. For the purposes of this report, we generally took as our starting point the IPCC definition, i.e., “The degree to which a system is susceptible to, or unable to cope with, adverse effects of climate change, including climate variability and extremes. Vulnerability is a function of the character, magnitude, and rate of climate variation to which a system is exposed, its sensitivity, and its adaptive capacity" (IPCC, 2007a). Most of what we define as "vulnerability indicators" in this report primarily encompass sensitivity and exposure to environmental stresses, and we do not focus on adaptive capacity. The indicators we discuss relate generally to the vulnerability of aquatic ecosystems, ecosystem services, and drinking water supplies.

Our first challenge was to identify guidelines for classifying the comprehensive suite of 623 indicators. The goal was to divide them into vulnerability indicators versus those indicators that merely measure the current state of a resource. The vulnerability indicators, at least in principle, could measure the degree to which the resource being considered (e.g., watershed, ecosystem, human population) is susceptible to, and unable to cope with, adverse effects of externally forced change. Such change potentially includes climate or any other global change stressor.

We determined that, in practical terms, the essence of a vulnerability indicator is that it should inherently include some kind of relative or value judgment, e.g., comparing one watershed to another, comparing it to some objectively defined threshold or possible state, or reporting on its change over time, as opposed to measuring water quality or ecological condition at a point in time without reference to anything else. Applying these criteria, we winnowed the original list of 623 indicators down to 53, and in the report we discuss the degree to which indicators from this reduced set might reflect vulnerability of water quality and aquatic ecosystems to challenges from long-term global change stresses.

8.1.2. Challenges Part II: Determining Relative Vulnerability

Determination of the relative vulnerability of a particular location using a given vulnerability indicator (or an index, if multiple indicators have been combined), can be accomplished by comparing the value of the indicator to a gradient of values measured at different locations. Alternatively, one can capitalize on objective vulnerability thresholds for some indicators. Such thresholds reflect abrupt or large changes in the vulnerability of water quality or aquatic ecosystems in response to a small change in a stressor, sometimes but not always associated with a particular regulatory threshold. Such thresholds are most useful when they distinguish between acceptable and unacceptable conditions.

We searched for thresholds for our 53 vulnerability indicators from three different categories: human health-based thresholds, ecological thresholds, and sustainability thresholds. In the literature, we most often encountered the use of arbitrary cutoffs to separate relative vulnerability categories (e.g., high, medium, and low). We were only able to map objective thresholds for a small subset of the indicators, though in some cases we suggested modification of an indicator definition to facilitate the identification of thresholds. The lack of available functional break points for most indicators is to be expected. Many indicators respond to stress linearly or along a gradual gradient. For others, objective break points may be characterized through additional research, either through meta-analysis of previous research efforts or through new data collection and analysis. Future research may also yield additional insights into how break points for some indicators vary spatially (Link, 2005).

8.1.3. Challenges Part III: Mapping Vulnerability

The effort to produce indicator maps for this report faced a number of classic cartographic challenges. Most of these challenges fell into the following two major categories: data availability and mappability, and spatial aggregation.

8.1.3.1. *Data and Mappability*

Data availability and suitability were the most serious limitations in evaluating whether or not we could produce maps for the 53 vulnerability indicators. Issues we encountered included the following:

- Lack of national coverage;
- Varying scales of the data;
- Varying duration of the data records;
- Multiple datasets needed to be combined;
- Extensive modeling effort was required to generate values for the indicator;
- No dataset available for the indicator; and
- Data collection was in progress.

These data availability and suitability issues were often identified during the literature review. For example, study authors sometimes explicitly noted the need for data for particular indicators that were potentially useful. In other cases, these issues emerged only after beginning the process of attempting to create maps. For example, the limited spatial extents of some datasets were identified during the mapping process. A major lesson we learned from this project was that it may often be impossible to establish mappability without beginning the process of manipulating and mapping the various datasets involved.

Overall, these data and mappability issues reduced the starting set of 53 vulnerability indicators to a set of 25 vulnerability indicators for which we were able to create example maps.

8.1.3.2. *Spatial Aggregation*

To create a national map for a given indicator of vulnerability, one must aggregate data collected at discrete locations and calculate summary statistics that describe conditions across a larger area, such as the mean value of an indicator or the percentage of sites that exceed a threshold value. As noted above, a major research gap is the lack of objective, functional thresholds between "vulnerable" and "not vulnerable" for most of the indicators we investigated. A complementary challenge is that, even if such functional breakpoints can be found, it may be difficult to aggregate in such a way that these breakpoints remain meaningful.

The major issues we encountered were the following:

- Local variation and spatial heterogeneity in data collection sites;
- The choice of spatial frameworks (e.g., watersheds, ecoregions, coasts); and
- The extent (resolution) of the spatial unit chosen.

As illustrated with a variety of example maps, these methodological choices can lead to very different results, and hence different conclusions about relative vulnerability in one location compared to another.

A systematic process for refining or re-defining indicators of vulnerability to account for the challenges summarized above is likely to be valuable. Such a process is presented in Figure

7-2. For example, the Acid Neutralizing Capacity (#1) indicator is defined as the ability of a stream to buffer acidic inputs from acid rain or acid mine drainage. This indicator can be refined to measure the percentage of sites that with ANC less than 100 millequivalents/L to account for the aggregation challenge. In addition, indicators can be refined to more explicitly incorporate the exposure component of vulnerability. If elements of environmental change, such as temperature or precipitation, can be explicitly incorporated into the indicator, then future changes in this indicator can be modeled using predicted changes in the values of these elements. This strengthens the ties between the indicator and changes that may occur in the future, and facilitates the generation of more useful forecasts for decision-makers.

8.1.4. Challenges Part IV: Combining Indicators

Ultimately, the value for global change assessments of a database of indicators, and their maps, rests in how they can be examined holistically. Such indicators and their maps can also be examined in combination with scenarios of changes in critical external stressors, such as climate and land use. We showed some simple examples of how one might use such scenario data to highlight locations around the country where, for example, we might see a convergence between an already stressed water supply system, a warmer, drier climate, and significant population growth. One of several more sophisticated approaches involves designing indicators that explicitly include a functional dependence on a stressor that is expected to change over time, such as temperature, precipitation, or population.

We also considered the challenges associated with compositing multiple indicators in some way and mapping the result. This brings up issues of determining the functional equivalency of the different levels of relative vulnerability measured by the very different indicators, with no absolute standard as an anchor point for weighting their contributions. Creation of a uniform scoring system (e.g., 1, for lowest, and 5 for highest, vulnerability) resolves the practical difficulties of mapping but not the conceptual ones of establishing the relative contribution of each indicator to overall vulnerability. Appendix H includes an evaluation of the effects of aggregation on the validity of theoretical breakpoints for each of the mapped indicators based on the process outlined in Figure 8-1.

A possible way forward is in the development of what we refer to as "vulnerability profiles," based on multivariate statistical analyses such as PCA. As a simple example, we conducted a PCA on the mapped indicators. The six principal components we extracted tended to be associated with different potential dimensions of vulnerability: i.e., PC1 with at-risk species; PC2 with streamflow availability and usage; PC3 with pesticides in surface water; PC4 with macroinvertebrates and stream habitat quality; PC5 with meteorological drought indices; and PC6 with herbicides in groundwater. This kind of analysis allows the identification of watersheds or other geographic units with similar vulnerability profiles. This has the potential to be useful in the transfer of successful management or adaptation strategies from one location to another.

This process can be used to evaluate and guide the modification of potential indicators. The questions are oriented around the definition of vulnerability and the suitability of the indicator for mapping. Appendix H provides an evaluation of each of the 25 mappable indicators within the framework of the five questions presented in this flowchart.

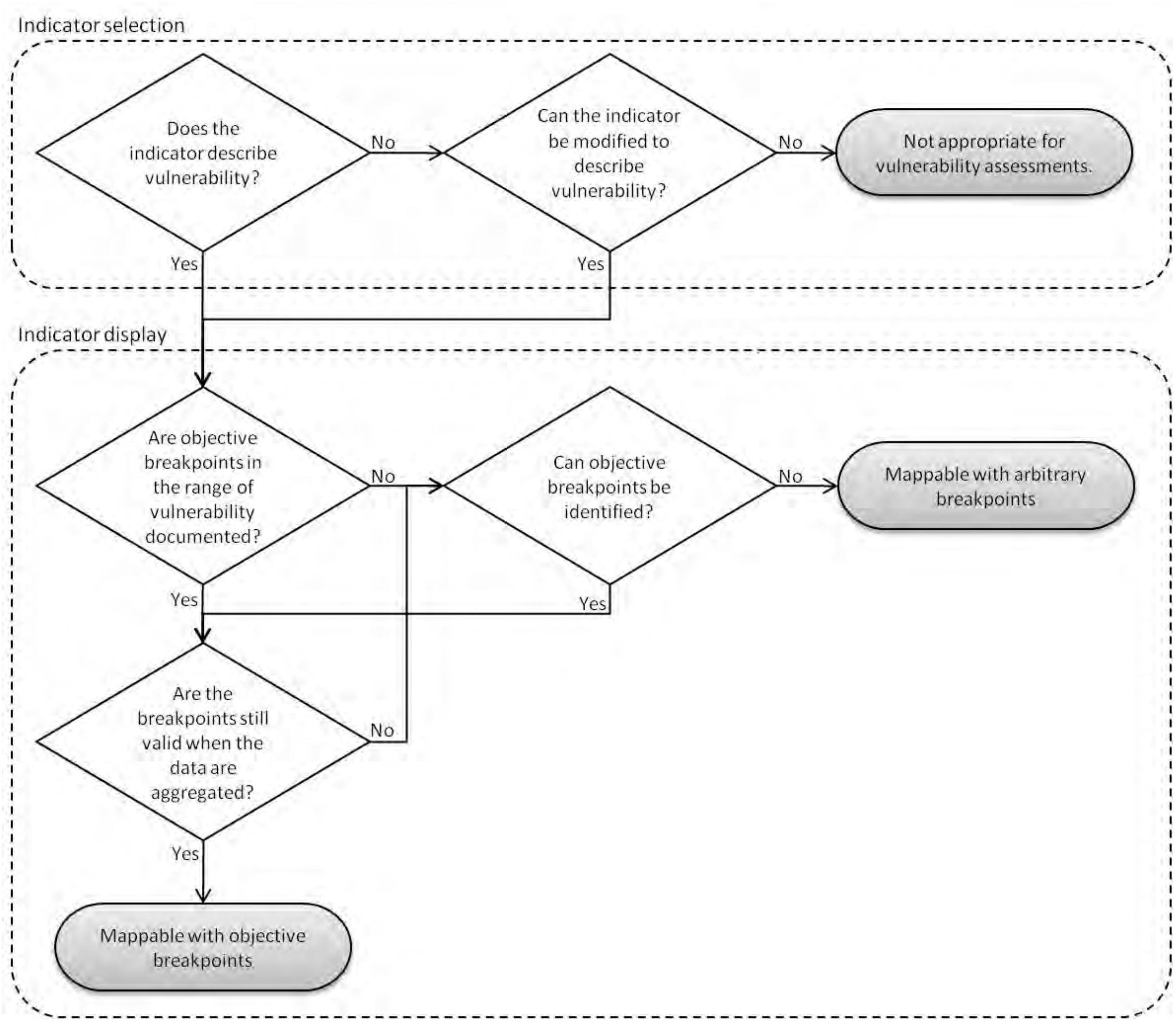


Figure 8-1. Indicator evaluation process.

8.2. RECOMMENDATIONS FOR FUTURE RESEARCH

As a result of exploring the challenges and issues described above, we have identified a number of areas where additional research is likely to contribute significantly to our ability to carry out indicator-based vulnerability assessments – both in the specific context of the indicators discussed in this report, and more generally.

8.2.1. Assessment of Non-mappable Indicators

Some indicators were designated as non-mappable due to the need for additional processing of available data, statistical analyses, evaluation of modeled data, or other tasks that were beyond the scope of this study. Enhanced modeling efforts that combine probabilistic (Bayesian) and mechanistic approaches may be particularly useful in defining minimum data collection requirements and for characterizing the interactions between physical, chemical, and biological processes. Additional effort to address these needs may yield highly useful maps of these indicators.

Examples of the data evaluation needs include:

- Acquiring and assembling national-scale wetland data: Wetlands may be significantly affected by climate and land-use change. Unfortunately, one important indicator for wetlands, Wetland Loss (#325), was designated as non-mappable, due to the effort required to download and process the data from the National Wetlands Inventory (NWI). The online ordering system requires users to download individual datasets at the 7.5 minute (1:24K) or 15 minute (1:100K) scales. In the lower 48 states, the USGS has designated approximately 56,500 1:24K-scale quadrangles. It may be possible to acquire national wetlands coverage from the U. S. Fish & Wildlife Service, and conduct subsequent analyses that would result in a national wetlands indicator.

- Assessment of the National Inventory of Dams database: Instream connectivity (#620) is an important measure that can be used to make inferences about drinking water availability (e.g. large reservoirs) and aquatic ecosystem functions (e.g. migration of species). To produce an accurate assessment of connectivity, it is important to have a comprehensive source of dam locations and diversions in the United States. The National Inventory of Dams, managed by the U.S. Army Corps of Engineers, is an attempt at such a data set, but some data (especially data pertaining to small dams) is absent from the database, available digital maps of the stream network are of varying quality and detail across the country, and the available data for dams are frequently inaccurate. An assessment of this database is needed and, if possible, additional dam data should be obtained to produce a map for this indicator. Work by the USGS on the National Hydrography Dataset and the NHD-plus is currently underway and should provide useful data in the coming years. A challenge to reporting this indicator will be evaluating what percentage of dams is omitted because they are too small to be registered in the national database on dams.

- Digitization and analysis of national flood plain data: The Population Susceptible to Flood Risk (#209) indicator evaluates the human population currently residing within a 500-year flood plain. A map for this indicator could be obtained by overlaying estimates of the 500-year flood plain from the Federal Emergency Management Agency (FEMA) with population data from the U.S. Census Bureau. However, according to FEMA's Map Service Center, GIS-compatible digital flood plain data were not available at the time of this study for several areas within the U.S. FEMA is currently working on a multi-year project to update and digitize national flood plain data. In the absence of a national flood plain data set, it would be useful to utilize existing digital flood plain data for urbanized areas to evaluate the percentage of metropolitan populations that may be prone to flooding.

8.2.2. Identifying Opportunities to Enhance Source Data

The indicators evaluated during this study were associated with data sets with varying degrees of completeness, ranging from large national assessment efforts, to indicators with no clear data source. Additional research is needed to identify opportunities to enhance the utility of national data sets and fill significant data gaps.

Examples of large national data sets that were used for this study include the EPA Wadeable Streams Assessment or the USGS National Water Quality Assessment (NAWQA) Program. These are unique data sets that yield high-quality data, but even these excellent data collection efforts fall short of providing the data density required to produce robust analyses of vulnerability over large scales, e.g., at the scale of a 4-digit HUC unit, as calculated values may be highly sensitive to a few or even a single measurement taken at a discrete location within the spatial aggregation unit. Additional research is needed to evaluate data collection effort required to enhance the statistical power of these key datasets.

In addition, some example maps produced for this study could be improved by addressing significant gaps in the source data. For example, the data set used to produce Instream Use / Total Streamflow (#351) did not include estimates of groundwater recharge, one of the input variables for this indicator, for some regions. For these regions, we assumed recharge was equal to withdrawals. The accuracy of this indicator in these areas would be improved by acquiring better estimates for the missing variable.

Furthermore, some data sets that are regularly updated through ongoing data collection activities may have quality problems. For example, the Centers for Disease Control and Prevention’s (CDC) Waterborne Disease and Outbreak Surveillance System (WBDOSS), a potential data set for the Waterborne Human Disease Outbreaks (#322) indicator, relies on voluntary reporting of water-related disease outbreaks by public health departments of U.S. states, territories, and local governments. The data are inconsistent and of variable quality. Ideally, data would be reported regularly for all parts of the country and consistently documented by a single responsible entity. Alternatively, if voluntary data collection by multiple entities continues, stringent guidelines might be set forth to ensure the quality of the data in this database.

Finally, some of the indicators that we deemed to be non-mappable because we could not identify any existing data source have the potential to be highly useful measures. Additional research to identify the data needed to calculate appropriate vulnerability metrics, collect new data, or transform existing data to calculate and map these indicators would be valuable.

8.2.3. Development of New Indicators from Available Data Sets

A direct follow-up effort to the methodology employed for this study would be a review of existing national-scale environmental data sets to determine which might lend themselves to the development of new, useful indicators. This would allow for more opportunities to create indicators that are specifically tailored to the needs of local planners and decision-makers. For example, a new indicator, Water Demand, defined as the total water withdrawals in millions of gallons per day, can be created based on data available from the USGS’ National Water-Use Data set. A map of this indicator is shown in Figure 5-3. Assessment of vulnerability using this indicator, perhaps in combination with indicators of water availability such as Groundwater Depletion (#121) and Net Streamflow per Capita (#623), may be useful at a variety of scales, from national to local, for understanding the water budgets of communities. This would facilitate responses with, for example, improved conservation policies in areas subject to severe water shortages.

Using available data as a starting point would also enhance our ability to work with indicators with objective thresholds that distinguish between acceptable and degraded condition. For example, in the present study a set of five pesticide indicators [#367, #369, #371, #373, and #374] were mapped using USGS' NAWQA data set. These indicators were designed by USGS to provide a cumulative assessment of multiple pesticides present in ambient water by calculating an average concentration. It is difficult to determine thresholds for these indicators given the diversity of pesticides and the varying levels of risks they pose. Instead, the development of new indicators for individual pesticides, using the same data set, would allow us to map the data using established thresholds, such as MCLs, to categorize vulnerability. Individual pesticide indicators may present regional patterns and identify regional water quality concerns, whereas the combined indicators developed by USGS and used in this study may mask local and regional vulnerability.

8.2.4. Need for Additional Study and Data Collection in Coastal and Other Areas

We note that the example indicators mapped for this study do not represent an even distribution across the possible categories of water quality and aquatic ecosystem vulnerability indicators. Our heavy focus on areas such as water quantity, freshwater ecosystems, and certain aspects of water quality is a result of the methodology applied, and not a reflection of bias on the part of the investigators or advisors selecting indicators and mapping data. Furthermore, as we have emphasized throughout the report, the selection of indicators that were mapped is not intended to imply anything about which indicators are inherently more important for assessing vulnerability to global climate change and other stressors. Rather, the example maps are for illustration of our methodology, and the selection of indicators for mapping was based on the ready availability of data.

Data on the location of streams and quantity of surface water flow were generally readily available in readily usable formats. There are several critical areas within the study of water quality and aquatic ecosystems, however, which suffer more than other areas from the challenges and data limitations discussed in this report. Additional research is needed in the areas of coastal aquatic systems, wetlands, freshwater tidal marshes, and the fish and animal habitats they support. Additional data collection over longer time periods and greater spatial extents is needed to capture the characteristics and trends in the condition and vulnerability of these important systems.

8.2.5. Use of Indicators for Future Studies

The focus of the present study was to identify indicators of water quality and aquatic ecosystem condition that represented vulnerability and could be mapped at the national scale. 598 indicators were eliminated from the original comprehensive list of indicators for various reasons that made them unsuitable for a national-scale vulnerability assessment. However, many of these indicators may be valuable for other studies or purposes.

Many indicators were eliminated because their associated data sets did not have comprehensive national coverage or may only be relevant in some areas. Although these indicators had limited utility for the present study, they are likely to be valuable for conducting vulnerability assessments at regional or local scales. For example, EPA National Coastal Assessment data for the Water Clarity Index [#318] and Water Quality Index [#319] indicators are only available for the Gulf coast region. Similarly, Snowpack Depth [#440] is only measured in regions where rivers and other surface water sources are primarily fed by snowmelt, such as in the Colorado River basin. Mangrove Cover [#63] is only relevant where these trees grow – a small portion of the Gulf Coast. Each of these indicators may be highly useful for monitoring changes over time in local systems and for guiding local decisions in response to observed or expected changes. A useful follow-up effort to this study would be the development of an indicator compendium that would describe the geographic extent and available data sources for indicators that are relevant at local and regional scales. Local decision makers could use this resource in conjunction with the national-scale indicators presented in this study to guide local planning efforts.

Indicators whose data were based on future projections were also eliminated because the present study only examined current vulnerability. For example, data for Heat-Related Illnesses Incidence [#392] are available as estimates of mortality from the National Center for Health Statistics (NCHS) based on three climate change scenarios for the years 2020 and 2050. Data for land cover or land use indicators, such as Coastal Wetlands (acreage) (#52) and Urban and Suburban Areas (acreage) (#308), Population susceptible to flood risk (#209), and other population-related indicators, may be projected into the future using output data from climate and earth system models. These data, while not useful for the present study, are useful in understanding future vulnerability, particularly when taking into account the effects of climate change on human and natural environments. Understanding future vulnerability is a crucial component of many ongoing and planned research studies aimed at strategic planning for adaptation to the effects of global climate change.

8.2.6. Establishment of Stress-response Curves, Vulnerability Thresholds, and Baseline Conditions

In this report we focused on the development of methods to assess relative vulnerability. Additional research to evaluate how individual indicators respond to stress (e.g., sensitivity, threshold response, resistance, etc.) will facilitate assessments of absolute vulnerability linked to system function. There is a large body of basic ecological and sociological research that will need to be created before this issue can be comprehensively addressed. The issue of thresholds, much discussed above, is of course intimately related.

Furthermore, observationally establishing baseline conditions, and implementing more routine monitoring for locally relevant indicators, would enable water resource managers to identify significant water quality and ecological changes over time, which would allow the development of additional indicators, or more accurate calculation of existing indicators, for assessment.

8.2.7. Drawing on other Established Approaches for Combining Indicators

In particular, a comparison of the traditional multivariate approaches for combining indicators to the approaches used by EPA’s ReVA program, such as the generalized weighted distance method, may be fruitful. Future research efforts could apply the ReVA aggregation methods to the indicators in this report, which are topically and spatially broader. Such aggregation would also allow relationships between components of vulnerability for the indicators specified in this study to be addressed. Future work could include the design of new, robust indicators using existing data sources.

8.2.8. Incorporating Landscape and Land Use Metrics

Landscape metrics, such as percent natural cover, roads crossing streams, and agriculture on slopes, can provide additional context for the indicators presented in the report. Metrics such as these may assist with the interpretation of sensitivity. Land use metrics that specify the sources of polluted runoff (e.g., urban areas, Concentrated Animal Feeding Operation areas) and of polluted groundwater (e.g., septic systems in low-lying areas) are useful for assessing the vulnerability of surface and subsurface water quality, respectively. Measurements of human impact may explain an indicator's vulnerability score or may suggest an alternative interpretation. In addition, some metrics, such as population growth rate, can be used to assess future exposure to stress (see, for example, Figure 7-1).

8.2.9. Incorporating Information Based on Remote Sensing Technologies

Remote sensing technologies have facilitated measurement of a variety of landscape and land use indicators. They are commonly used to measure fragmentation of forests, the influence of urbanization and suburbanization on the landscape, and for quantification of land cover / land use categories (e.g., how the extent of forests or croplands have changed over time). Remote sensing can also be used to investigate how local ecologies have been disturbed by human encroachment. Remote sensing is currently being employed for the measurement of chlorophyll a and turbidity.

8.2.10. Incorporating Metrics of Adaptive Capacity

Vulnerability to future changes depends in part on choices made by society today and into the future. In the context of climate change in particular, adaptive capacity is the ability of an ecosystem or society to continue to perform its range of functions despite changes in factors that affect those functions. A system has inherent adaptive capacity when its natural attributes make it resilient to stress, whereas institutional adaptive capacity includes policies, practices, and infrastructure that create options for meeting human and ecosystem needs in the face of an uncertain future. The specific attributes or actions that create adaptive capacity are largely different for aquatic life and human uses of water, although there is some overlap among these categories.

Differentiating inherent and institutional adaptive capacity is useful because it points to two different management approaches. Systems with inherent adaptive capacity are less vulnerable, even when they are sensitive and exposed to stress. Thus, many advocate directing planning and management efforts toward systems lacking this capacity. Institutional adaptive capacity can be built in many ways (for examples, see IPCC, 2007a). Many of these strategies require a significant shift from short to long term planning, which is typically resisted by institutional and infrastructural inertia. Many specific practices involve diversification and the creation of redundancy, which can be hard to justify in the context of current conditions. Some also require acknowledgement of fundamental uncertainty about the future.

Community-based analyses have shown that the conditions that interact to shape exposures, sensitivities, adaptive capacities, and hence create needs and opportunities for adaptation, are community-specific (Smit and Wandel, 2006). This finding suggests that any attempt to transfer adaptive strategies among regions must look for commonalities both in the magnitude of vulnerability and in its qualitative, multi-dimensional profile. As described above, some of the techniques described in this report (e.g., the development of vulnerability profiles and similarity maps) could, in principle, be used to identify such commonalities among regions, which, in combination with case studies of successful adaptation, would provide guidance for potential policy transfer, or serve as a screening tool for the feasibility of adaptive strategy transfer.

As we said above, we hope that this report will be a useful building block for future work on multi-stressor global change vulnerability assessments. Ultimately, we believe the work described here is a preliminary contribution toward bridging disconnects between the decision support needs of the water quality and aquatic ecosystem management communities and the priorities and capabilities of the global change science data and modeling communities; to the synthesis of insights across more detailed, place-based, system-based, or issue-based case studies (e.g., in individual watersheds, wetlands, urban ecosystems) to obtain national-scale insights about impacts and adaptation; and to prioritization of future work in developing adaptation strategies for global change impacts.

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